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METHODOLOGY

Method for Estimation of Physical Impacts and Monetary Valuation for Priority Impact Pathways

Part 1: Impact Assessment

Part 2: Economic Valuation

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PART I: IMPACT ASSESSMENT

EXECUTIVE SUMMARY

1. INTRODUCTION
1.1 Background To The ExternE Project 1
1.2 Previous Studies 2
1.3 Objectives and Scope of the ExternE Project 3
1.4 Purpose and Structure of the Report 5
1.5 References 6

2. THE IMPACT PATHWAY METHODOLOGY 7
2.1 Approaches Used for Externality Analysis 7
2.2 Guiding Principles in the Development of the ExternE Methodology 11
2.3 Defining the Boundaries of the Analysis 11
2.4 Analysis of Impact Pathways 16
2.5 Impacts Assessed in the ExternE Project 27
2.6 Summary 28
2.7 References 30

3 MODELLING POLLUTANT TRANSPORT AND CHEMISTRY 31
3.1 Introduction 31
3.2 Atmospheric Transport Models 32
3.3 Considerations on the Appropriate Range of Analysis 34
3.4 Local Scale Modelling of Primary Pollutants 36
3.5 Regional Scale Modelling of Primary Pollutants and Acid Deposition 39
3.6 Ozone Modelling 46
3.7 Conclusions 54
3.8 References 56

4. PUBLIC HEALTH EFFECTS OF AIR POLLUTION ARISING FROM FOSSIL FUEL COMBUSTION 61
4.1 Introduction 61
4.2 Acute Effects of PM, SO₂, NOₓ: Model Assumptions 67
4.3 Acute Effects of Ozone 85
4.4 Chronic Effects 103
4.5 Summary of Model Assumptions 108
8. IMPACTS OF ACIDIC DEPOSITION AND PHOTO-OXIDANTS ON AGRICULTURE 226
8.1 Effects of Pollutants on Agriculture 226
8.2 European Crop Production 233
8.3 Exposure-Response Functions 236
8.4 Assessment of Impacts of Acidification on Agricultural Soils 245
8.5 Assessment of Impacts of Nitrogen Deposition on Agricultural Soils 246
8.6 Pollutant x Pest Interactions 247
8.7 Conclusions 247
8.8 References 249

9. IMPACTS OF ACID RAIN AND OZONE ON FORESTS 254
9.1 Introduction 254
9.2 Pollution Effects on Forest Health and Recent Declines 257
9.3 Available Models of Tree Response to Air Pollution 261
9.4 Alternative Approaches 268
9.5 Conclusions 279
9.6 References 281

10. EFFECTS OF ACIDIFICATION ON RECREATIONAL FISHERIES 286
10.1 Introduction 286
10.2 Linking Hydrochemical and Fish Status Models 289
10.3 Mitigation Costs 294
10.4 Summary 295
10.5 References 297

11. IMPACTS OF AIR POLLUTION ON BUILDING MATERIALS 300
11.1 Introduction 300
11.2 Impact Pathways of Fossil Fuel Cycle Pollutants on Materials 301
11.3 Assessment of Damage to Building Materials 307
11.4 Identification of Dose-Response Functions 316
11.5 Estimation of Impacts on Materials 330
11.6 Conclusions 335
11.7 References 337
Appendix 11.1 Building Identikits used for the UK and German Implementations 342
EXECUTIVE SUMMARY

This report describes the methodology used by the ExternE Project of the European Commission (DGXII) JOULE Programme for assessment of the external costs of energy. It is one of a series of reports describing analysis of nuclear, fossil and renewable fuel cycles for assessment of the externalities associated with electricity generation. Part I of the report deals with analysis of impacts, and Part II with the economic valuation of those impacts.

Analysis is conducted on a marginal basis, to allow the effect of an incremental investment in a given technology to be quantified. Attention has been paid to the specificity of results with respect to the location of fuel cycle activities, the precise technologies used, and the type and source of fuel. The main advantages of this detailed approach are as follows:

- It takes full and proper account of the variability of impacts that might result from different power projects;
- It is more transparent than analysis based on hypothetically ‘representative’ cases for each of the different fuel cycles;
- It provides a framework for consistent comparison between fuel cycles.

A wide variety of impacts have been considered. These include the effects of air pollution on the natural and human environment, consequences of accidents in the workplace, impacts of noise and visual intrusion on amenity, and the effects of climate change arising from the release of greenhouse gases. Wherever possible we have used the ‘impact pathway’ or ‘damage function’ approach to follow the analysis from identification of burdens (e.g. emissions) through to impact assessment and then valuation in monetary terms. This has required a detailed knowledge of the technologies involved, pollutant dispersion, analysis of effects on human and environmental health, and economics. In view of this the project brought together a multi-disciplinary team with experts from many European countries and the USA.

The spatial and temporal ranges considered in the analysis are dependent on the type of impact under assessment. For example, noise effects will only be experienced over a maximum of a few kilometres; impacts associated with emission of acidifying pollutants from power stations act over one thousand kilometres or more; impacts associated with long lived radioisotopes emitted to the atmosphere, or greenhouse gases, require global assessment. In all cases we have sought to quantify impacts over as much of the range affected as possible.

Similar variation between impacts exists with respect to the timescales involved - some impacts are short lived, others will persist for many thousands of years. The assessment of long-term impacts brings in a need for discounting. The central discount rate used here is 3%, though a rate of 0% has also been used for transparency, to demonstrate the problem of discounting long term impacts, and also to demonstrate sensitivity to discount rate.
This report aims to present the methodology in a transparent manner in more detail than has been possible elsewhere, clearly identifying the uncertainties involved and the assumptions that are being made. A major difficulty concerns the quantitative description of the uncertainties associated with the analysis. The major problem is that some aspects of uncertainty cannot be described numerically from the available data. To date a largely qualitative assessment of uncertainty has been adopted although new approaches are currently being investigated within ExternE. These will be incorporated into the methodology during the next phase of the Project.

Some observers have stated that the detailed methodology described in this report imposes unreasonable analytical demands. However, ExternE and other recent projects have demonstrated that this is no longer the case. Software developed under ExternE will soon be available, further reducing the demands placed upon analysts. The methodology described in this report thus provides a means of providing externality data that incorporates the latest scientific and economic knowledge on the external costs of energy which can then be incorporated within decision making processes.
1. INTRODUCTION

1.1 Background to the ExternE Project

The European Commission launched the project in collaboration with the US Department of Energy in 1991. The study was initiated because of concern that the environmental impacts of energy use were not being properly integrated into decision making processes. Advances in scientific knowledge over recent years have demonstrated that the use of energy causes damage to a wide range of receptors, including human health, natural ecosystems and the built environment. These damages are referred to as external costs, or externalities, to the extent that they are not reflected in the market price of the good in question (in this case energy). Concern has been heightened because of a realisation that the damage that has been observed or predicted as a consequence of current activities will be irreversible in some cases, and for others may persist for many years, and be widespread.

Work on externalities relating to the energy sector is being driven by several factors, including:

- The need to integrate environmental concerns when choosing between different fuels and energy technologies;
- The need to evaluate the costs and benefits of stricter environmental standards;
- Increased attention to the use of economic instruments for environmental policy;
- Different policy initiatives to encourage competition and the market mechanism in the energy sector (e.g. privatisation, limiting of subsides, liberalisation of energy markets).

At the level of the European Union the need for externality assessment has been stressed by a number of measures, including:

- The Maastricht Treaty;
- The Fifth Environmental Action Programme ‘Towards Sustainability’;
- The Commission's White Paper entitled ‘Growth, competitiveness, employment and ways forward to the 21st century’ (CEC, 1993);
- The proposal for an energy-carbon tax;
- The requirement of the European Environmental Agency ‘to stimulate the development of methods of assessing the cost of damage to the environment and the costs of environmental preventive, protection and restoration policies’.

Further discussion on the requirements for externality assessment in these measures is provided in the ExternE Project’s Summary Report (European Commission, 1995a).
Concern over the environmental impacts of energy are of course not confined to Europe. Similar developments are apparent in a number of other countries such as the USA (partly through collaboration with the ExternE Project), Australia, Canada and the former EFTA countries, and international organisations including the International Energy Agency, United Nations Economic Commission for Europe and the International Atomic Energy Agency.

1.2 Previous Studies

Previous work in this area can be divided into those studies performed specifically to quantify external costs, and those which have concentrated on the quantification of impacts without necessarily continuing to quantification of economic damage.

Two broad methodological approaches have been used in these studies. They are usually described as ‘top-down’ and ‘bottom-up’. Top-down analyses use highly aggregated data, for example, national emission and impact data, to estimate the damage costs of particular pollutants. The main problem with this technique is that it tends to make poor use of the wealth of environmental and economic knowledge that is now available. Also, the approach is not suited to the calculation of marginal costs, which are of most interest in neo-classical economic analysis. The advantage of the technique is in some ways a reflection of its problems; it has low data requirements and may allow preliminary estimates to be made where detailed information is lacking.

In contrast the bottom-up methodology (also known as the damage function or impact pathway approach) allows the use of technology-specific emissions data for individual locations. This is the approach selected for the ExternE Project. It uses detailed site and technology specific data together with pollution dispersion models, detailed information on the location of receptors and dose-response functions to calculate the physical, chemical biological and social impacts of fuel cycle activities. These impacts are then valued in monetary terms.

The most prominent of past externality studies are those by Hohmeyer (1988), Ottinger et al (1990) at Pace University, Bernow et al (1990) of the Tellus Institute, ECO Northwest (1987) and Pearce et al (1992). The Tellus Institute study differs to the others in that it used the control cost method to quantify damages, and hence no attempt was made to quantify impacts separately from their valuation. The other studies show a progression from fairly restricted top-down analysis to the more complete fuel cycle based bottom-up approach. The merits of these studies are discussed in more detail in the introduction to the ExternE Project Summary Report (European Commission, 1995a). One of their main criticisms is that much of the work is derivative, with each ‘new’ study reliant on information from earlier studies. Furthermore, the analysis has tended to be dominated by economic arguments, with less attention being devoted to the merits of the science underlying the impact assessment.
1.3 Objectives and Scope of the ExternE Project

1.3.1 Objectives

The ExternE Project aims to be the first systematic approach to the evaluation of external costs of a wide range of different fuel cycles.

The principle objectives of the ExternE Project, to date, have been:
1. To develop a unified methodology for quantifying the environmental impacts and social costs associated with production and consumption of energy;
2. To use this methodology to evaluate the external costs of incremental use of different fuel cycles in different locations in the European Union;
3. To identify critical methodological issues and research requirements.

The emphasis in the objectives has been on methodology rather than calculation of values of external costs. This reflects concerns, particularly in the USA, that values for external costs are quoted and used without reference to the assumptions and methodology, both of which have a major influence on many calculations.

The ExternE Project has been built on three important principles. First, transparency, to show how the work is done, the assumptions made and the data used. Secondly, comprehensiveness, to include all significant impacts and extending the impact analysis in time and space to capture all relevant effects, and thirdly, consistency, to allow comparisons between different fuel cycles and different locations. The project has critically reviewed and used the best available scientific data, models and dose-response functions in the literature.

The design of internationally agreed tools is a necessary first stage for estimating reliable values for external environmental costs. Quantitative estimates of some external costs have been made to demonstrate the methodology developed in the study and to compare against other studies. Hence, the objective has been to advance the state of the art, rather than to produce definitive results, and identify areas which require further research work.

Work on the ExternE Project is to continue under the European Commission’s 4th Framework Programme. It is envisaged that future work will build on the existing framework but will focus more on the potential application of externalities in policy and decision making.

1.3.2 Scope

The project addresses complete ‘cradle-to-grave’ analyses for site and technology specific fuel cycles. For power generation it takes into account the chain of processes from fuel extraction and processing, to power generation and waste disposal, including normal and accidental operation, and for some fuel cycles electricity transmission. The impacts related to the use of the construction materials and transport of bulk materials and personnel are also taken into account if estimated to be an important source of externalities (for example, in some renewable cycles). In these cases, the approach is close to a life cycle analysis. The analysis
is done on a marginal basis, such that external costs are calculated for a new incremental
investment in each fuel cycle activity.

The methodology that has been developed could be applied to all energy systems. However,
to date the project has focused on the important power generation fuel cycles (fossil, nuclear
and renewables) and energy saving. Some work is also underway on the application of the
methodology to evaluate the externalities associated with domestic energy use and the energy-
related externalities of transport.

In this project, the term environmental externalities is broadly defined and includes all
burdens imposed by an activity on the environment that affect our welfare. Hence it includes
impacts of pollution on human health, agriculture, materials, ecosystems and how the
resultant changes in ecosystems affect our actual, potential or future possibilities to use it
(recreation, use for transportation) or the importance we may attach to conserving it
(biodiversity). Externalities may include both negative economic effects (e.g. damages) and
positive economic effects (benefits) on the environment and health. Externalities of energy
are of course not limited to environmental and health related impacts; impacts on employment
and energy security also give rise to (non-environmental) externalities. Although this project
has been defined to cover a number of these aspects, to date work has focused on the
environmental and health impacts.

The ExternE Project goes beyond earlier studies in several respects. These include:

- A more thorough characterisation of the energy technologies and their discharges into the
  environment on a site specific basis;
- Consideration of all major stages of fuel cycles rather than just electric power generation.
  Significant environmental impacts occur during the mining of fuels, their transportation
  and the eventual disposal of the wastes. These need to be evaluated as carefully as those of
  the generating stage but previously little work has been done on them;
- Modelling the dispersion and transformation of pollutants rather than relying on previous
  estimates. Existing work has relied mainly on short distance dispersion models for
  assessment of air pollution impacts. Concern over the transboundary effects of pollution,
  and subsequent international agreements demonstrate that it is widely recognised that this
  is inadequate. Furthermore, attention needs to be paid to the transformation of pollutants,
  leading, for example, to the generation of sulphate aerosols or ozone.
- Engaging in a more extensive, critical review and use of the literature on ecology, health
  sciences and economics than previous studies. Many ecological, epidemiological and
  valuation studies have come out in the last five to ten years. The ExternE Project has
  reviewed these and used the latest information available in developing impact pathways
  and in valuing them.
- Identifying cases where externalities may be partially, or wholly, internalised.

The ExternE Project study is closely linked with other work being undertaken within the
European Commission's programmes. For example, an overall objective of this project is that
the results of this project will provide the environmental dimension in the new generation of
energy-economy-environment models (Systems for Externalities and General Equilibrium,
SOLFEGE Project), currently under development within the JOULE programme. The
internalisation of external costs into energy policies requires not only a better assessment of the external costs but also tools to assess different policy options and to identify the optimal mix of instruments for internalising external costs. The combination of the framework developed within the ExternE Project with the new generation of energy-economy-environment models will provide a set of powerful tools well suited to tackle such policy issues.

1.4 Purpose and Structure of the Report

The objective of the present report is to provide details of the broad methodology developed and used by the ExternE Project for impact assessment. In addition to this, background information is also presented on a number of specific impact pathways to provide a forum for detailed discussion and to prevent a need for repetition in other volumes in the present series. Most of these pathways concern the impacts of emissions of the acid gases, particulates and ozone precursors to the air, and hence are primarily associated with the fossil fuel cycles. Detailed descriptions of the application of the methodology to the different fuel cycles considered in the project are given in other ExternE Project reports (European Commission, 1995b-e).

The next chapter provides details of the impact pathway approach. This is followed by a review of pollutant dispersion modelling, and then assessment of impacts on human health, either as it affects members of the public through air pollution or accidents, or workers, through occupational diseases and accidents. The following five chapters discuss the methodology for air pollution effects on agriculture, natural terrestrial ecosystems, forests, freshwater fisheries and building materials. Chapter 12 discusses the methodology for global warming, Chapters 13 addresses noise and Chapters 14 and 15 address visual intrusion and major accidents.

Within this report relatively little attention is devoted to assessment of nuclear and certain aspects of renewable fuel cycles. In each case the impacts concerned tend to be quite specific to the technologies involved, and hence these issues have been dealt with in detail in the ExternE reports on individual fuel cycles (European Commission, 1995d and 1995e). Part II of this report contains details of the methodologies used for the economic valuation of the impacts.
1.5 References


ECO Northwest, (1987). Generic Coal Plant Study: Quantification and Valuation of Environmental Impacts, for Bonneville Power Administration, Portland, OR.


2. THE IMPACT PATHWAY METHODOLOGY

2.1 Approaches Used for Externality Analysis

The ExternE Project has adopted the ‘impact pathway’ approach for the assessment of the external impacts and associated costs resulting from the supply and use of energy. The phrase ‘impact pathway’ simply relates to the sequence of events linking a ‘burden’ to an ‘impact’ and subsequent valuation. The methodology therefore proceeds sequentially through the pathway, as shown in Figure 2.1. It provides a logical and transparent way of quantifying externalities. However, only recently, through developments in environmental science and economics, and improvements in computing power has it become a realistic proposition. Subsequently a number of other approaches have been used in the past. These alternatives are now reviewed to highlight the differences that exist between them and the ExternE methodology.

2.1.1 The ‘top-down’ approach

Early externalities work used a ‘top-down’ approach (the impact pathway approach is ‘bottom-up’ in comparison, although the labels "macro" and "micro" might be more descriptive). A top-down (macro) analysis is typically highly aggregated, being carried out at a regional or national level, using estimates of the total quantities of pollutants emitted or present and estimates of the total damage that they cause. A classic example is the study by Hohmeyer (1988), followed in the US by Ottinger et al (1991). The key steps of Hohmeyer's analysis of fossil power plants were as follows:

- Development of an inventory of emissions of carbon monoxide (CO), particulate matter (PM), oxides of nitrogen (NOx), sulphur dioxide (SO2), and volatile organic compounds (VOCs) in Germany;
- Weighting of these emissions by relative toxicity factors;
- Estimation of the contribution of fossil power plants to the total damage from these pollutants;
- Review of the available literature on estimates of environmental damage due to air pollution and extraction of a plausible range of values for damages in Germany, for the major impact categories of flora, fauna, mankind, materials, and climate change;
- Combination of the numbers to obtain damage costs per kWh of electricity, for each of these five impact categories.

This scheme is summarised in Figure 2.2. While Hohmeyer's study was a pioneering step forward, there are a number of major problems with it. Even if one can trust the damage cost estimates, the relative toxicity factors are a weak link because they are derived from government regulations for maximum permissible concentrations at a place of work, rather than from exposure-response functions.
Figure 2.1 An illustration of the main steps of the impact pathways methodology applied to the consequences of pollutant emissions. Each step is analysed with detailed process models.
2.1.2 The ‘control cost’ method

The ‘control cost’ method substitutes the cost of reducing emissions of a pollutant (which are determined from engineering data) for the cost of damages due to these emissions. As justification, the proponents of this approach argue that when elected representatives decide to adopt a particular level of emissions control they are expressing the collective willingness-to-pay of the society that they represent to avoid a damage. However, that would be true only if our society were at the economic optimum. In fact, the point of the optimum is precisely what we would like to know, and the control cost approach merely begs the question. Certainly, knowledge of control costs is an important element in formulating prescriptive regulations, but presenting them as if they were damage costs is to be avoided.

2.1.3 Life cycle analysis (LCA)

Life cycle analysis (OECD, 1992; Heijungs et al, 1992; Lindfors et al, 1995) is a flourishing discipline whose roots go back to the net energy analyses that were popular twenty years ago. While there are several variations, all life cycle analysis is in theory based on a careful and
holistic accounting of all energy and material flows associated with a system or process. The approach tends to be used to compare the environmental impacts associated with different products that perform similar functions, such as plastic and glass bottles.

One of the major issues in any LCA concerns the way in which the ‘system boundaries’ are specified, defining the extent of the analysis around the basic system under investigation. Some limit on the system is clearly essential, otherwise analysis becomes impracticable. However, much thought needs to go into the drawing of boundaries in order that all factors relevant to the outcome of the analysis are assessed. The ‘system’ should therefore include flows that are induced upstream or downstream of the processes under investigation.

The Society for Environmental Toxicology and Chemistry (SETAC) has sought to define an internationally agreed methodology for LCA, in order to overcome certain problems that have been identified, and to allow comparison between the results of different studies (Consoli et al., 1993). The SETAC methodology proceeds through the following stages:
1. Definition of objectives and system boundaries;
2. Collation of data to provide an emissions inventory;
3. Classification of emissions into impact categories;
4. Characterisation of impacts;
5. Normalisation;
6. Valuation (translation of impact assessments into a common unit).

Stage 4, ‘characterisation’ typically only goes as far as summing emissions of various pollutants using some kind of weighting scheme (e.g. global warming potentials). Several systems for ‘characterising’ different types of pollutant have been collated by Heijungs et al. (1992). The following stage, normalisation, consists of an assessment of the relative weighting to be given to different impact categories. The term valuation refers to assessment of environmental effects to a common unit, though this is not necessarily monetary.

Life cycle analyses tend not to be explicit on the calculation of impacts, if they have attempted to quantify impacts at all. For example, the ‘classification factors’ identified by Heijungs et al. (1992) for each pollutant are independent of the site of release. For air pollution these factors were calculated with the assumption of uniform mixing in the earth's atmosphere. While this can be justified for greenhouse gases and other pollutants with long residence times, it is unrealistic for particulate matter, NOx, SO2 and ozone (O3). The reason for this radical approximation lies in the choice of emphasis in LCA: the prime goal is a careful accounting of all material flows, direct and induced. Since induced flows occur at many geographically different points under a variety of different conditions, it is simply not practicable to keep track of the local details of the emissions.

By contrast, the fuel cycle analysis reported here places its emphasis on the quantification of impacts and cost because people care more about impacts than emissions. The quantification of emissions is merely a step in the analysis. From this perspective the choice between the
approaches taken by fuel cycle analysis and by conventional life cycle analysis is a matter of accuracy; uncertainties increase the further the analysis is continued. In general terms, however, it is our view that the fuel cycle analyses of the ExternE Project can be considered a particular example of life cycle analysis.

2.2 Guiding Principles in the Development of the ExternE Methodology

The underlying principles on which the methodology for the ExternE Project has been developed are:

*Transparency*, to show precisely how results are calculated, the uncertainty associated with the results and the extent to which the external costs of any fuel cycle have been fully quantified.

*Consistency*, with respect to the boundaries placed on the system in question, to allow valid comparisons to be made between different fuel cycles and different types of impact within a fuel cycle.

*Analysis is conducted on a marginal basis*, to allow the results to be used to assess the incremental effects and costs of (e.g.) investment in new power projects or changes in government policy.

In order to comply with these principles, the ExternE analysis presented in this series of reports is based around the assessment of the effects of individual power projects which are closely specified with respect to:

- The technologies used;
- The location of the power generation plant;
- The location of supporting activities;
- The type of fuel used;
- The source and composition of the fuel used.

Each of these factors is important in determining the magnitude of impacts and hence associated externalities.

2.3 Defining the Boundaries of the Analysis

The starting point for fuel cycle analysis is the definition of the boundaries of the system under investigation, and the range of burdens and impacts to be addressed. The boundaries used in the ExternE Project are very broad. This is essential in order to ensure consistency in the application of the methodology for different fuel cycles.

Certain impacts brought within our boundaries cannot be quantified at the present time, and hence the analysis is incomplete. However, this is not a problem peculiar to this style of analysis; it simply reflects on gaps in available knowledge. Our rule here is that no impact that is known or suspected to exist, but cannot be quantified, should be ignored for convenience. Instead it should be retained for consideration alongside whatever analysis has been possible. An advantage of the present analysis is that such gaps have been identified. Further work is needed so that unquantified effects can be better integrated into decision making processes.
2.3.1 Stages of the fuel cycle

For any project associated with electricity generation the system is centred on the generation plant itself. However, the system boundaries should be drawn so as to account for all potential effects of a fuel cycle. In our analysis we have included a wide variety of other activities. The exact list of stages is clearly fuel cycle dependent (the list for the nuclear fuel cycle will be very different to that for wind). The following provides examples of the up- and down-stream processes that we have considered. Reference should be made to the reports on individual fuel cycles to see what has been included in each case:

- Production of construction materials;
- Transport of construction materials;
- Construction of plant;
- Exploration for fuel;
- Extraction of fuel;
- Processing of fuel;
- Transport of fuel;
- Transport of personnel;
- Treatment of flue gases;
- Generation of wastes and by-products (‘wastes’ that can be used elsewhere);
- Further treatment of waste;
- Removal of plant at the end of its service lifetime;
- Restoration of sites after closure.

In practice, a complete analysis of each stage of a fuel cycle is often not necessary in order to meet the objectives of the analysis (see below). However, the onus is on the analyst to demonstrate that this is the case - it cannot simply be assumed. Worth noting is the fact that variation in laws and other local conditions will lead to major differences between the importance of different stages in different parts of the world; one should not ignore a particular stage simply because someone elsewhere found that it was not important.

A further complication arises because of the linkage between fuel cycles and other activities, upstream and downstream. For example, in theory we should probably account for the externalities associated with (e.g.) the production of materials for the construction of the plant used to make the steel that is used to make turbines, coal wagons, etc. One could carry on to infinity. The benefit of doing so is, however, extremely limited. Fortunately this can usually be demonstrated quite easily through order-of-magnitude calculations on emissions, without the need for detailed analysis (see Section 2.4.1, below).

The treatment of waste matter and by-products deserves special mention. Impacts associated with waste sent for disposal should be considered as part of the system under analysis. However, impacts associated with waste utilised elsewhere (by-products) should be considered as part of the utilising system from the moment that they are removed from the boundaries of the fuel cycle. For these purposes the fuel cycle boundary may need to extend to treatment of wastes to ensure that they are in a form that can be used elsewhere. A good example is the use of gypsum produced through flue gas desulphurisation as a building material.
2.3.2 Location of fuel cycle activities

One of the distinguishing features of the present study is the inclusion of site dependence. For each stage of each fuel cycle we have therefore identified specific locations for the power plant and all of the other activities drawn within the system boundaries. In some cases this has gone so far as to identify routes for the transport of fuel to power stations. The reason for defining our analysis to this level of detail is simply that location is important in determining the size of impacts. There are several elements to this, the most important of which are:

- Variation in legal requirements (e.g. concerning the use of pollution abatement techniques, occupational safety standards, etc.);
- Variation in fuel quality;
- Differences in the sensitivity of the human and natural environment upon which fuel cycle burdens impact.

We have therefore set out to identify realistic sites for each of the technologies considered in each country for which analysis has been conducted. Some of the sites are currently used for, or in support of, electricity generation. Others are not, though they are suitable to be used in the way assumed. For example, neither the West Burton ‘B’ or Lauffen plants considered in the coal fuel cycle report (European Commission, 1995a) currently exist. However, planning permission was granted to the CEGB (the UK’s electricity utility at the time) to build a coal fired plant at West Burton in 1988, and the Lauffen site was identified as suitable for a large fossil fuel power station in a survey conducted in Germany.

The alternative to this would be to describe a ‘representative’ site for each activity. It was agreed at an early stage of the ExternE study that such a concept is untenable. Also, recent developments elsewhere, such as use of critical loads analysis in the revision of the Sulphur Protocol within the United Nations Economic Commission for Europe’s (UNECE) Convention on Long Range Transboundary Air Pollution, demonstrate the importance attached to site dependence by decision makers.

However, the selection of a particular series of sites for a particular fuel cycle is not altogether realistic, particularly in relation to upstream impacts. For example, how can one evaluate the externalities associated with oil used for a power plant when the power station uses oil imported from a number of different countries, and that oil serves a number of different end uses (gasoline, heating oil, plastics, etc.) as well as fuel oil for the power plant? The assumption of a single site certainly simplifies the analysis, but can hardly be considered realistic, other than in certain cases such as plants burning lignite and some coal fired plants.

In principle the basic rule for the analysis is simple:

\[
\text{Incremental upstream impact of the fuel cycle} = \text{difference of impacts of world fuel production with and without this fuel cycle.}
\]

However, application of this rule is anything but straightforward. More sophisticated treatment will become possible in the future, as the analysis is applied to more and more cases (provided that this is done in a consistent fashion), establishing a more extensive database on externalities.
than is currently available. In view of this, the current phase of ExternE should be regarded as a first step, upon which to build future work.

2.3.3 Identification of fuel cycle technologies

The objective of the ExternE Project so far has been to quantify the external costs of power generation technologies built in the 1990s. For the most part we have not been concerned so far with future technologies that are as yet unavailable, nor with older technologies which are gradually being decommissioned.

Over recent years an increasingly prescriptive approach has been taken to the regulation of new power projects. The concept of Best Available Technology/Techniques (BAT), coupled with emission limits and environmental quality standards defined in Europe by both national and international legislation, restrict the range of alternative plant designs and rates of emission. This has made it relatively easy to select technologies for each fuel cycle on a basis that is consistent across fuel cycles. However, care is still needed to ensure that a particular set of assumptions are valid for any given country. This is illustrated by the different NO\textsubscript{x} emission standards applied in the UK and Germany (see Chapter 2 of the report on the coal fuel cycle - European Commission, 1995 - in the present series).

Whilst the present reports deal with closely specified technology options, extrapolation of the results to other options can often be achieved with little additional effort. Details of emissions from 8 technology options, including cogeneration of heat and power, are given in the report on the coal fuel cycle, though the report itself only goes on to provide detailed analysis for 2 of these. Externalities can be quantified for the other options by direct extrapolation based on the specified emissions data, assuming certain similarities (e.g. location and stack height).

This section has outlined the broad strategy used to define technologies so far by the ExternE Project. For further details the reader should consult the reports on the individual fuel cycles.

2.3.4 Identification of fuel cycle burdens

For the purposes of this project the term ‘burden’ relates to anything that is, or could be, capable of causing an impact of whatever type. The following broad categories of ‘burden’ have been identified:

- Solid wastes;
- Liquid wastes;
- Gaseous and particulate air pollutants;
- Accidents;
- Occupational exposure to hazardous substances;
- Noise;
- Heat;
- Presence of human activity (causing, e.g., visual intrusion);
- Others (e.g. exposure to electro-magnetic fields).
During the identification of burdens no account has been taken of the likelihood of any particular burden actually causing an impact, whether serious or not. For example, in spite of the concern that has been voiced in recent years there is no definitive evidence that exposure to electro-magnetic fields associated with the transmission of electricity is capable of causing harm. The purpose of the exercise is simply to catalogue everything to provide a basis for the analysis of different fuel cycles to be conducted in a consistent and transparent manner, and to provide a firm basis for revision of the analysis as more information on the effects of different burdens becomes available in the future.

The need to describe burdens so comprehensively is highlighted by the fact that it is only recently that the effects of long range transport of acidic pollutants, and the release of CFCs and other greenhouse gases have been appreciated. Ecosystem acidification, global warming and depletion of the ozone layer are now regarded as among the most important environmental concerns facing the world. The possibility of other apparently innocuous burdens causing risks to health and the environment should not be ignored.

2.3.5 Identification of impacts

The next part of the work involves identification of the potential impacts of these burdens. At this stage it is irrelevant whether a given burden will actually cause an appreciable impact; all potential impacts of the identified burdens should be reported.

The emphasis here is on making the analyst demonstrate that certain impacts are of little or no concern, according to current knowledge. The conclusion that the externalities associated with a particular burden or impact, when normalised to fuel cycle output, are likely to be negligible is an important result. It will not inevitably follow that action to reduce the burden is unnecessary, as the impacts associated with it may for example have a serious effect on a small number of people. It does, however, imply that the use of an externality ‘adder’ to electricity price would be too blunt an instrument to deal with the burden efficiently.

The present series of reports provides comprehensive listings of burdens and impacts for most of the fuel cycles considered. The tasks outlined in this section and the previous one are therefore not as onerous as they seem, and will become even easier with the development of appropriate databases.

2.3.6 Valuation criteria

Many receptors that may be affected by fuel cycle activities are valued in a number of different ways. For example, forests are valued not just for the timber that they produce, but also for providing recreational resources, habitats for wildlife, their interactions (direct and indirect) with climate and the hydrological cycle, protection of buildings and people in areas subject to avalanche, etc. Externalities analysis should include all such aspects in its valuation. Again, the fact that a full quantitative valuation along these lines is rarely possible is besides the point when seeking to define what a study should seek to address.

2.3.7 Spatial limits of the impact analysis
The system boundary also has spatial and temporal dimensions that require definition. The spatial limits of analysis should be designed to capture impact as fully as possible. Within the ExternE Project the objective has been to quantify impacts over their full range, irrespective of national boundaries. The importance of this is illustrated in the next Chapter by the demonstration that fuel cycle pollutants can be transported over great distances. In practice it is frequently necessary to truncate the analysis at some point, because of limits on the availability of data. It is recommended that an estimate be provided of the extent to which the analysis has been restricted. For example, one could quantify the proportion of emissions of a given pollutant that have been accounted for, and the proportion left unaccounted.

The importance of an adequate definition of spatial limit for the analysis is illustrated by European Commission (1995a) in the analysis of health impacts associated with emissions from coal-fired power plants. Originally the range of this analysis was restricted to a distance of 50 km around the emission site. This was increased to the national boundary in the UK and Germany, and then again to a full European assessment for the German case study. A highly significant increase in estimated damages was found each time the range was extended.

2.3.8 Temporal limits of the impact analysis

In keeping with the previous section, impacts should be assessed over their full time course. This clearly introduces a good deal of uncertainty for long term impacts, such as those of global warming or high level radioactive waste disposal, as it requires a view to be taken on the structure of future society. There are a number of facets to this, such as global population and economic growth, technological developments, the sustainability of fossil fuel consumption and the sensitivity of the climate system to anthropogenic emissions.

The approach adopted here is that discounting should only be applied after impacts are quantified. The application of any discount rate above zero can reduce the cost of major events in the distant future to a negligible figure. This perhaps brings into question the logic of discounting over time scales running far beyond the experience of recorded history.

We believe that it is informative to conduct analysis of impacts that take effect over periods of many years in spite of the large uncertainties involved. By doing so we can at least gain some idea of how important they might be in comparison to effects experienced over shorter time scales. We can also identify the chief methodological and ethical issues that need to be addressed. To ignore them might suggest that they are unlikely to be of any importance.

2.4 Analysis of Impact Pathways

Having identified the range of burdens and impacts that result from a fuel cycle, and defined the technologies under investigation, the analysis typically proceeds as follows:

- Prioritisation of impacts;
- Description of priority impact pathways;
- Quantification of burdens;
- Description of the receiving environment;
- Quantification of impacts;
2.4.1 Prioritisation of impacts

It is possible to produce a list of several hundred burdens and impacts for many fuel cycles. A comprehensive analysis of all of these is clearly beyond the scope of externality analysis. What is important, however, is to be sure that the analysis covers those effects that (according to present knowledge) will provide the greatest externalities. Accordingly, the analysis presented in this series of reports was limited, though only after due consideration of the potential magnitude of all impacts that were identified for any fuel cycle. Wherever possible scoping calculations were made to gain some idea of the likely magnitude of impacts, during the identification of the priority impacts.

It is necessary to ask whether the decision to assess only a selection of impacts in detail reduces the value of the project as a whole. We believe that it does not, as it can be shown that many impacts (particularly those operating locally around any given fuel cycle activity) will be negligible compared to the overall damages associated with the technology under examination.

There are good reasons for believing that local impacts will tend to be of less importance than regional and global effects. The first is that they tend to affect only a small number of people. Even though it is possible that some individuals may suffer very significant damages these will not amount to a significant effect when normalised against a fuel cycle output in the order of several Tera-Watt hours per year. It is likely that the most appropriate means of controlling such effects is through the local planning system, which should have the flexibility to deal effectively with the wide range of concerns that may exist locally.

A second reason for believing that local impacts will tend to be less significant is that it is typically easier to ascribe cause and effect for impacts effective over a short range than for those that operate at longer ranges. Accordingly there is a longer history of legislation to combat local effects. It is only in recent years that the international dimension of pollution of the atmosphere and water systems has been realised, and action has started to be taken.

There are obvious exceptions to the assertion that local impacts are of less importance than others, as will be evident from the following sections. The most important probably concerns occupational disease and accidents that affect workers and members of the public. Given the high value attached to human life and well-being there is clear potential for associated externalities to be large. Other cases mainly concern the renewable technologies for which the most serious impacts, particularly of the power generation stage, tend to be extremely localised. For example, most concern over the development of wind farms typically relates to visual intrusion in natural landscapes and to noise emissions.

The analysis of certain upstream impacts appears to create difficulties for the consistency of the analysis. For example, if we treat emissions of SO\textsubscript{2} from a power station as a priority burden, why not include emissions of SO\textsubscript{2} from other parts of the fuel cycle, for example from the production of the steel and concrete required for the construction of the power plant? In
conjunction with our colleagues in the US Department of Energy study, we have assessed a number of such cases using available databases, such as GEMIS (Fritsche et al, 1992). Calculations made in this way typically show that the emissions under investigation are 2 or 3 orders of magnitude lower than those from a fossil fuel power station. It is thus logical to expect that the impacts of such emissions are trivial in comparison, and can safely be excluded from the analysis. However, this does not hold across all fuel cycles. In our report on the wind fuel cycle (European Commission, 1995b), for example, it was found that emissions associated with the manufacture of plant are capable of causing significant externalities, relative to the others that were quantified.

The selection of priorities partly depends on whether one wants to evaluate damages or externalities. In quite a few cases the externalities are small in spite of significant damages. For example, if a power plant has been in place for a long time, much of the externality associated with visual and noise impacts will have been internalised through adjustments in the local real estate market; the impacts become reflected in the price of housing. It has been argued that occupational health effects are also likely to be internalised. For example, if coal miners are rational and well informed their work contracts should offer benefits that internalise the incremental risk that they are exposed to. However, this is a very controversial assumption, as it depends precisely upon people being both rational and well informed and the existence of perfect mobility in labour markets. For the present time we have quantified occupational health effects in full, leaving the assessment of the degree to which they are internalised to a later date.

It would be wrong to assume that those impacts given low priority in this study are always of so little value from the perspective of energy planning that it is never worth considering them in the assessment of external costs. Each case has to be assessed individually. Differences in the local human and natural environment, and legislation need to be considered.

Section 2.5 provides a listing of the priority impact categories for the fossil, nuclear and renewable fuel cycles considered so far by ExternE.

2.4.2 Description of priority impact pathways

Some impact pathways analysed in the present study are relatively simple in form, as illustrated in Figure 2.3. This Figure uses the example of externalities associated with visual intrusion arising from the development of a wind farm. The scheme is so simple that diagrammatic representation is really unnecessary.
In other cases the link between ‘burden’ (defined here simply as something that causes an ‘impact’) and monetary cost is far more complex. To clearly define the linkages involved in such cases we have drawn a series of diagrams. One of these is shown in Figure 2.4, illustrating the series of processes that need to be accounted for from emission of acidifying pollutants to valuation of impacts on freshwater ecosystems.

A number of points should be made about Figure 2.4. It (and others like it) do not show what has been carried out within the project. Instead they illustrate an ideal - what one would like to do if there was no constraint on data availability. They can thus be used both in the development of the methodology and also as a check once analysis has been completed, to gain an impression of the extent to which the full externality has been quantified. This last point is important because much of the analysis presented by the ExternE Project is incomplete. This reflects on the current state of knowledge of the impacts addressed, rather than on the efforts of the study team! The analysis can easily be extended once further data becomes available.

Also, for legibility, numerous feedbacks and interactions are not explicitly shown in the diagrammatic representation of the pathway.
**Figure 2.4** The impact pathway showing the series of linkages between emission of acidifying pollutants and the valuation of impacts to freshwater fisheries. This complex example contrasts with the simple scheme shown in Figure 2.3.
2.4.3 Quantification of burdens

The data used to quantify burdens must be both current and relevant to the situation under analysis. Emission standards, regulation of safety in the workplace and other relevant factors vary significantly over time and between and within different countries. These differences are important, and are recognised by the specificity of our analysis with respect to time and place.

It is true that the need to meet these demands creates difficulties for data collection. However, given that the objective of this work is to provide as far as possible an accurate account of the environmental and social burdens imposed by energy supply and use, these issues should not be ignored.

All impacts associated with pollution of some kind require the quantification of emissions. Emission rates of the ‘classical’ air pollutants (CO₂, SO₂, NOₓ, CO, volatile organic compounds and particulate matter) are quite well known. Especially well determined is the rate of CO₂ emission for fuel using equipment; it depends only on the efficiency of the equipment and the carbon/hydrogen ratio of the fuel - uncertainty is negligible. Emissions of the other classical air pollutants are somewhat less certain, particularly as they can vary with operating conditions. For example, NO emissions are likely to increase above manufacturer’s specifications if a selective catalytic reduction unit is not well maintained. The sulphur content of different grades of oil and coal can vary by an order of magnitude, and hence, likewise, will emissions unless this is compensated for through varying the performance of abatement technologies. The general assumption made in this study is that unless otherwise specified, the technology used is the best available according to the regulations in the country of implementation, and that performance will not degrade. We have sought to limit the uncertainty associated with emissions of these pollutants by close identification of the source and quality of fuel inputs within the study.

The situation is less clear with respect to trace pollutants such as lead and mercury, since the content of these in fuel can vary by much more than an order of magnitude. Furthermore, some of these pollutants are emitted in such small quantities that even their measurement is difficult. The dirtier the fuel, the greater the uncertainty in the emission estimate. There is also the need here to account for emissions to more than one media, as trace pollutants may be emitted to air, contained in water discharged to rivers or the sea, or entrained in material sent for disposal or use on land. The last category is the subject of major uncertainty, as waste has historically been sent for disposal to facilities of varying quality, ranging from simple holes in the ground to well-engineered landfills. Increasing regulation relating to the disposal of material and management of landfills should reduce uncertainty in this area greatly for analysis within the European Union, particularly given the concept of self-sufficiency enshrined in Regulation 259/93 on the supervision and control of shipments of waste into, out of and within the European Community. The same will not apply in many other parts of the world.

The problem becomes more difficult for the upstream and downstream stages of the fuel cycle because of the variety of technologies that may be involved. Particularly important may be some stages of fuel cycles such as biomass, where the fuel cycle is potentially so diverse that it is possible that certain activities are escaping stringent environmental regulation.
The burdens discussed so far relate only to routine emissions. Burdens resulting from accidents also need to be considered. These might result in emissions (e.g. of oil) or an incremental increase in the risk of injury or death to workers or members of the public. Either way it is normally necessary to rely upon historical data to quantify accident rates. Clearly the data should be as recent as possible so that the rates used reflect current risks. Major uncertainty however is bound to present when dealing with extreme events, such as the disasters at Chernobyl and on the Piper Alpha oil rig in the North Sea. To some extent it is to be expected that accident rates will fall over time, drawing on experience gained.

Wherever possible data should be relevant to the country where a particular fuel cycle activity takes place. For example, it is not logical to use UK and German occupational health data for coal mining for an implementation in a country which does not use coal from either country, perhaps preferring to import from non-EU countries, such as Poland, the USA, China or South Africa. Although this complicates the analysis, it is simply a reflection of the real world.

2.4.4 Description of the receiving environment

The use of the impact pathway approach requires a detailed definition of the scenario under analysis with respect to both time and space. This includes:

- Meteorological conditions affecting dispersion and chemistry of atmospheric pollutants;
- Location, age and health of human populations relative to the source of emissions;
- The status of ecological resources;
- The value systems of individuals.

The range of the reference environment for any impact requires expert assessment of the area influenced by the burden under investigation. As stated above, arbitrary truncation of the reference environment is methodologically wrong and will produce results that are incorrect. It is to be avoided as far as possible. Guidance can be taken from the existing series of fuel cycle reports (European Commission, 1995a, b, c, d).

Clearly the need to describe the sensitivity of the receiving environment over a vast area (extending to the whole planet for some impacts) creates a major demand on the analyst. This is simplified by the large scale of the present study - we have been able to draw on data held in many different countries. Further to this we have been able to draw on numerous databases that are being compiled as part of other work, for example on critical loads mapping. Databases describing the distribution of the key receptors covering the whole of Europe are currently under development as part of the ExternE Project software.

Some assumption about future activities is required in order to estimate damages at some time in the future. In a few cases it is reasonable to assume that conditions will remain roughly constant, and that direct extrapolation from the present day is as good an approximation as any. In other cases, involving for example the emission of acidifying gases or the atmospheric concentration of greenhouse gases this assumption is untenable. In the first case, most European countries are committed to reducing acid emissions under the UN ECE Convention on Long Range Transboundary Air Pollution. The area of Europe subject to exceedance of critical loads will thus decline over time. Estimates of future emissions and of their effect on
deposition over Europe are available, however, and can be included in the modelling framework. In the second case the atmospheric concentration of greenhouse gases is certain to increase for the foreseeable future, in spite of any commitment to freezing or reducing emission levels by any European countries. However, projected global emissions of greenhouse gases are available from the IPCC under six different scenarios for the future to the year 2100 (Pepper et al., 1992), and the effect of these emissions on global atmospheric greenhouse gas levels may be modelled. The scenarios also include a variety of other parameters required for assessment of the long impacts of climate change, such as population and economic growth. Unfortunately it is to be noted that the scenarios cover a very wide range of ‘possible futures’, underlining the uncertainties associated with conducting analysis over long time scales.

2.4.5 Quantification of impacts

The methods used to quantify various types of impact form the bulk of this report. As suggested by the impact pathways illustrated in Figures 2.3 and 2.4, the complexity of this analysis varies greatly. In some cases externalities can be calculated by multiplying together as few as 3 or 4 parameters. In others it is necessary to use a series of sophisticated models linked to large databases.

Common to all of the analysis conducted on the impacts of pollutants emitted from fuel cycles are the needs for modelling the dispersion of pollutants and the use of a dose-response function of some kind. Again, there is much variation in the complexity of the models used.

The most important pollutant transport models used so far within ExternE relate to the atmospheric dispersion of pollutants. These are addressed in Chapter 3 of this report. They need to account not only for the physical transport of pollutants by the winds but also for chemical transformation. A major problem has so far been the lack of a regional model of ozone formation and transport within fossil-fuel power station plumes that is applicable to the European situation. However, models are available for the other atmospheric pollutants of concern.

The term ‘dose-response’ is used somewhat loosely in much of this work, as what we are really talking about is the response to a given exposure of a pollutant in terms of atmospheric concentration, rather than an ingested dose. Hence the terms ‘dose-response’ and ‘exposure-response’ should be considered comparable. A major issue with the application of dose-response functions concerns the assumption that they are transferable from one context to another. For example, many of the functions for health effects of air pollutants are derived from studies in the USA. Is it valid to assume that these can be used in Europe? The answer to this question is to a certain degree unknown - there is good reason to suspect that there will be some variation, resulting from the affluence of the affected population, the exact composition of the cocktail of pollutants that the study group was exposed to, etc. However, in most cases the view of our experts has been that transference of functions is to be preferred to ignoring particular types of impact altogether - neither option is free from uncertainty.

Dose-response functions come in a variety of functional forms, some of which are illustrated in Figure 2.5. They may be linear or non-linear and contain thresholds (e.g. critical loads) or not.
Those describing effects of various air pollutants on agriculture have proved to be particularly complex, incorporating both positive and negative effects, because of the potential for certain pollutants, e.g. those containing sulphur and nitrogen to act as fertilisers.

![Graph showing different forms of dose-response functions](image)

**Figure 2.5** A variety of possible forms for dose-response functions.

Ideally these functions and other models are derived from studies that are epidemiological - assessing the effects of pollutants on real populations of people, crops, etc. This type of work has the advantage of studying response under realistic conditions. However, results are much more difficult to interpret than when working under laboratory conditions, where the environment can be closely controlled. One of the main problems with laboratory studies has been the need to expose study populations to extremely high levels of pollutants, often significantly greater than they would be exposed to in the field. Extrapolation to lower, more realistic levels may introduce significant uncertainties, particularly in cases where there is reason to suspect that a threshold may exist.

The description and implementation of exposure-response relationships is fundamental to the entire ExternE Project. Much of this report is, accordingly devoted to assessment of the availability and reliability of these functions.

**2.4.6 Economic valuation**

The rationale and procedures underlying the economic valuation applied within the ExternE Project are discussed in Part II of this report.
2.4.7 Assessment of uncertainty

Causes of uncertainty

Uncertainty in externality estimates arises from a number of sources, including:
- The variability inherent in any set of data;
- Extrapolation of data from the laboratory to the field;
- Extrapolation of exposure-response data from one geographical location to another;
- Assumptions regarding threshold conditions;
- Lack of detailed information with respect to human behaviour and tastes;
- Political and ethical issues, such as the selection of discount rate;
- The need to assume some scenario of the future for any long term impacts;
- The fact that some types of damage cannot be quantified at all.

It is important to note that some of the most important uncertainties listed here are not associated with technical or scientific issues, instead they relate to political and ethical issues, and questions relating to the development of world society. It is also worth noting that, in general, the largest uncertainties are those associated with impact assessment and valuation, rather than quantification of emissions and other burdens.

By contrast to the relatively small uncertainties and normal (gaussian) frequency distributions typically encountered in science and engineering, the uncertainties in impact analysis are so large that it would be inappropriate to use error intervals that are additively symmetric about the mean. Instead one should specify multiplicative intervals, in other words, intervals that are additive on a logarithmic scale. The frequency distributions are not symmetric, with implications that may appear counterintuitive. It is helpful to think in terms of lognormal distributions because they are frequently encountered in impact analysis, analogous to the normal distributions so familiar in the more exact sciences. A variable $x$ has a lognormal distribution if the variable $\ln(x)$ has a normal distribution; in other words, it is normal on a logarithmic scale.

Analogous to the ordinary normal (also known as gaussian) distribution which is characterised by two parameters, the mean and the standard deviation, the lognormal distribution can be characterised by the geometric mean and the geometric standard deviation $\sigma_G$. For this distribution the geometric mean is equal to the median: half of the distribution is above, the other half below the median. The geometric standard has a simple interpretation in terms of the 67% confidence interval (a familiar number because for gaussian distributions 67% of all values are within one standard deviation of the mean): for a lognormal distribution 67% of the values are within the interval $[1/\sigma_G, \sigma_G]$. Likewise 95% are within the interval $[(1/\sigma_G)^2, (\sigma_G)^2]$. Note, however, that these values are centred around the median rather than the mean; the lognormal distribution is not symmetric.
Methods for describing uncertainty

Traditional statistical techniques would ideally be used to describe the uncertainties associated with each of our estimates, to enable us to report a median estimate of damage with an associated probability distribution. Unfortunately this is rarely possible without excluding some significant aspect of error, or without making some bold assumption about the shape of the probability distribution. Alternative methods are therefore required, such as sensitivity analysis, expert judgement and decision analysis.

So far we have mainly relied upon the use of sensitivity analysis and expert judgement. For the sensitivity analysis we have sought to identify the most uncertain parameters in the analysis and then varied them over what we regard as a reasonable range. No attempt has been made to investigate the possible propagation of uncertainties through the impact pathways using sensitivity analysis - it has simply been applied to individual parameters.

Expert judgement has been used to report the level of uncertainty that we perceive is associated with each set of results, taking into account all of the uncertainties that may be present. Four confidence bands have been defined:

- H (high) - maximum uncertainty perceived to be less than an order of magnitude;
- M (medium) - maximum uncertainty perceived to be about an order of magnitude;
- L (low) - result may be in error by more than an order of magnitude;
- ! - reserved for particularly uncertain results (e.g. those of the analysis of global warming impacts) which could be in error by two or more orders of magnitude.

The confidence bands are extremely broad in their range, partly because of the types of damage that are being assessed, and partly because the use of expert judgement becomes somewhat meaningless if the bands are defined too tightly.

Future work on description of uncertainty

Clearly, none of the methods used so far to quantify uncertainty are ideal. An improved methodology is certainly required to allow the proper integration of uncertainty into decision making processes. This has already started to be developed, and further work is planned for the next stage of the ExternE Project.

The development work performed so far has concentrated on the most important externalities identified by the project. For each stage of the analysis the full range of possible outcomes needs to be defined. A probability distribution has then to be described within this range. The results for each stage can be combined using commercially available software. Further demonstration and discussion of this approach is necessary before it can be widely implemented.
2.5 Impacts Assessed in the ExternE Project

2.5.1 Priority impacts for fossil technologies

The following list of priority impacts has been derived for the fossil fuel cycles. It is necessary to repeat that this list is compiled for the specific fuel cycles considered by the present study, and should be reassessed for any new cases. The first group of impacts are common to all fossil fuel cycles:
1. Effects of atmospheric pollution on human health;
2. Accidents affecting workers and/or the public;
3. Effects of atmospheric pollution on materials;
4. Effects of atmospheric pollution on crops;
5. Effects of atmospheric pollution on forests;
6. Effects of atmospheric pollution on freshwater fisheries;
7. Effects of atmospheric pollution on unmanaged ecosystems;
8. Impacts of global warming;

To these can be added a number of impacts that are fuel cycle dependent:
10. Impacts of coal and lignite mining on ground and surface waters;
11. Impacts of coal mining on building and construction;
12. Resettlement necessary through lignite extraction;
13. Effects of accidental oil spills on marine life;
14. Effects of routine emissions from exploration, development and extraction from oil and gas wells.

2.5.2 Priority impacts for nuclear technologies

The priority impacts of the nuclear fuel cycle to the general public are radiological and non-radiological health impacts due to routine and accidental releases to the environment. The source of these impacts are the releases of materials through atmospheric, liquid and solid waste pathways.

Occupational health impacts, from both radiological and non-radiological causes, were the next priority. These are mostly due to work accidents and radiation exposures. In most cases, statistics were used for the facility or type of technology in question. When this was not possible, estimations were taken from similar type of work or extrapolated from existing information.

The impacts on the environment of increased levels of natural background radiation due to the routine releases of radionuclides have not been considered as a priority impact pathway. The most important impacts to the natural environment that could be expected would be the result of major accidental releases. This type of impact has been included in the economic damage estimates as the loss of land-use and agricultural products after a potential severe reactor accident. Possible long-term ecological impacts have not yet been considered.
2.5.3 Priority impacts for renewable technologies

The priority impacts for renewables vary considerably from case to case. Each case is dependent upon the local conditions around the implementation of each fuel cycle. For the wind fuel cycle (European Commission, 1995b) the following have been considered:
1. Accidents affecting the public and/or workers;
2. Effects on visual amenity;
3. Effects of noise emissions on amenity;
4. Effects of atmospheric emissions related to the manufacture of turbines and construction and servicing of the site.

Whilst for the hydro fuel cycle (European Commission, 1995b) we have assessed another group:
1. Occupational health effects;
2. Employment benefits and local economic effects;
3. Impacts of transmission lines on bird populations;
4. Damages to private goods (forestry, agriculture, water supply, ferry traffic);
5. Damages to environmental goods and cultural objects.

2.5.4 Related issues

A notable inclusion for the hydro fuel cycle was that of employment benefits. Clearly this is a consequence of all fuel cycles. However, its assessment and integration with other fuel cycle effects are the subject of much debate. Within the framework of the present study it was considered most appropriate to investigate this issue for only a single fuel cycle for the time being, in order to raise issues of concern.

It is necessary to ask whether the study fulfils its objective of consistency between fuel cycles, when some impacts common to a number of fuel cycles have only been considered in a select number of cases. In part this is due to the level of impact to be expected in each case - if the impact is likely to be large it should be considered. If it is likely to be small it may be legitimate to ignore it, depending on the objectives of the analysis. In general we have sought to quantify the largest impacts because these are the ones that are likely to be of most interest.

2.6 Summary

This Chapter has introduced the ‘impact pathway’ methodology of the ExternE Project. We believe that it provides the most appropriate way of quantifying externalities because it enables the use of the latest scientific and economic data.

Critical to the analysis is the definition of fuel cycle boundaries, relating not only to the different stages considered for each fuel cycle, but also to the:
- Location of each stage;
- Technologies selected for each stage;
- Identified burdens;
- Identified impacts;
• Valuation criteria;
• Spatial and temporal limits of impacts.

In order to achieve consistency it is necessary to draw very wide boundaries around the analysis. The difficulty with successfully achieving an assessment on these terms is slowly being resolved through the development of software and databases that greatly simplify the analysis.

The definition of ‘system boundary’ is thus broader than is typically used for LCA. This is necessary because our analysis goes into more detail with respect to the quantification and valuation of impacts. In doing so it is necessary to pay attention to the site of emission sources and the technologies used. We are also considering a wider range of burdens than is typical of LCA work, including, for example, occupational health effects and noise.

It will be noted that the analysis cuts across a large number of disciplines, ranging from energy technology to ecology and economics. In order to present a view that is consistently at or close to the current state of the art in each field it is necessary to use the skills of a wide variety of experts. Unless such an approach is adopted it is likely that the results (both quantitative and qualitative) would be rapidly outdated or prone to misinterpretation.

Perhaps the most important part of the analysis concerns the description of uncertainty. The best that we have been able to achieve at the present time is a semi-quantitative error analysis. However, we have used this to define broad confidence bands around each of our estimates. These are an important part of the results and should, accordingly, be given due consideration. An improved methodology for describing uncertainty for this type of analysis is currently being developed within the project.
2.7 References


3 MODELLING POLLUTANT TRANSPORT AND CHEMISTRY

3.1 Introduction

This chapter discusses the approaches to atmospheric modelling used within the ExternE Project. The atmospheric pollutants generated from the fuel cycles of the fossil fuels are predominantly emitted from the tall stacks of power plants. Therefore, we have concentrated on both primary emissions and pollutants associated with acidic deposition and photo-oxidants arising from fuel combustion. Although other sources will arise from up- and downstream fuel cycle phases, including emissions from construction activities, fuel extraction and transportation, these sources are generally negligible when compared to fossil fuel power station emissions.

For the nuclear fuel cycle, potential emissions of radionuclides will arise from many fuel cycle stages. We have not discussed the atmospheric processes and modelling of these species separately within this chapter. To model these radionuclide emissions, we have used standard modelling tools which assess stable atmospheric releases. The transport and modelling of all fuel cycle releases to other environmental media are not considered in this chapter; an in depth discussion of such processes and the methods used for modelling emissions to water, and through soil, can be found in the companion nuclear report (European Commission, 1995a).

The environmental impacts caused by burning fossil fuels in power stations are described in subsequent chapters. The primary emitted pollutants responsible for these impacts are listed below in Table 3.1. Many of the emissions are precursors for secondary pollutants, generated by chemical reactions in the atmosphere and so all compounds are listed, together with the impact categories for subsequent chapters.

This chapter does not address the modelling of fossil fuel cycle greenhouse gas emissions (CO₂, CH₄ and N₂O); our approach to global warming is presented in Chapter 12. Other atmospheric releases, including the large quantities of water vapour released from cooling towers are also not considered here. These sources may cause local effects but are likely to be better dealt with by local planning regulations rather than by integration of externalities into planning processes.

Due to the widespread nature of atmospheric pollution, it is rarely possible to deduce source-receptor relationships from measurements. Therefore, mathematical models including atmospheric dispersion, transformation and loss of pollutants are widely used to assess the consequences of present-day and predicted changes in emission patterns. These models vary in complexity and type in response to the perceived nature of the problem, and the range of
atmospheric behaviour observed. The accuracy of these models is limited by our understanding of atmospheric transport, by the chemical mechanisms involved and by the accuracy with which these processes can be represented on the best available computers. Restrictions on models may also be caused by inadequate or missing data. Despite these limitations, even quite simple models of pollutant behaviour have often been successful in explaining much of the variation in measured pollutant concentrations.

**Table 3.1** Primary and Secondary Pollutants from the Fossil Fuel Cycles.

<table>
<thead>
<tr>
<th>Primary (Emitted)</th>
<th>Secondary</th>
<th>Impact category</th>
</tr>
</thead>
<tbody>
<tr>
<td>CO₂ (Carbon dioxide)</td>
<td>-</td>
<td>Global warming</td>
</tr>
<tr>
<td>CH₄ (Methane)</td>
<td>O₃ (ozone)</td>
<td>Photo-oxidants, Global warming</td>
</tr>
<tr>
<td>N₂O (Nitrous oxide)</td>
<td>-</td>
<td>Global warming</td>
</tr>
<tr>
<td>SO₂ (Sulphur dioxide)</td>
<td>H₂SO₄</td>
<td>Health effects</td>
</tr>
<tr>
<td></td>
<td>Sulphate aerosol</td>
<td>S Fertilisation, Acid deposition, Global warming</td>
</tr>
<tr>
<td>NO (Nitric oxide)</td>
<td>NO₂</td>
<td>Health effects</td>
</tr>
<tr>
<td></td>
<td>HNO₃</td>
<td>N Fertilisation</td>
</tr>
<tr>
<td></td>
<td>Nitrate aerosol</td>
<td>Acid deposition</td>
</tr>
<tr>
<td></td>
<td>O₃</td>
<td>Photo-oxidants, Global warming</td>
</tr>
<tr>
<td>Particulates</td>
<td>-</td>
<td>Health effects, Deposition (soiling)</td>
</tr>
<tr>
<td>HCl (Hydrochloric acid)</td>
<td>NH₄Cl</td>
<td>Acid deposition</td>
</tr>
</tbody>
</table>

### 3.2 Atmospheric Transport Models

Atmospheric pollutants are transported by wind and diluted by atmospheric turbulence until they are deposited to the ground by either turbulent diffusion (dry deposition) or precipitation (wet deposition). Following emission from the stack, some of these primary pollutants take part in chemical reactions in the atmosphere to form secondary pollutants, such as sulphuric acid or ozone. The concentrated release of large quantities of pollutants (mainly oxides), from elevated point sources several hundred meters above the ground, leads to the specific behaviour of power station emissions. These differ in both dispersion and chemistry from widespread emissions released near ground level, for example by traffic and private households.

The atmospheric pollutant transport processes we have modelled in our analysis of fossil fuel cycles can be classified into three groups. These are separated according to their chemical characteristics and the atmospheric chemical and physical processes involved in their formation. They are:
• Primary pollutants directly emitted from the stack. These include particulate matter and sulphur dioxide (SO₂);
• Secondary sulphur and nitrogen species formed from the primary emissions of SO₂ and NOₓ. Analysis of these compounds includes modelling the concentration of secondary particulates in the atmosphere and dry and wet (acid rain) deposition processes;
• Photochemical oxidants, such as ozone, formed in atmospheric chemical reactions between hydrocarbons and oxides of nitrogen in the presence of sunlight.

For each of the above categories, a different modelling approach may be required. The first group, which comprises primary pollutants, are in effect chemically stable in the region of the emission. Thus, their concentrations can be predicted using Gaussian plume dispersion models. These models assume source emissions are carried in a straight line by the wind, mixing with the surrounding air both horizontally and vertically to produce pollutant concentrations with a normal (or Gaussian) spatial distribution. However, the use of these models is typically constrained to within a distance 100 km of the source.

Pollutant transport extends over much greater distances. A different approach is needed for assessing regional transport as chemical reactions in the atmosphere become important. This is particularly so for the acidifying pollutants. For this analysis we have used receptor-orientated Lagrangian trajectory models. The outputs from the trajectory models include atmospheric concentrations and deposition of both the emitted species and secondary pollutants formed in the atmosphere.

Finally, the impacts of photochemical formation from primary emissions must be considered. The modelling of the effects on ozone formation from power station emissions is difficult. Primary emissions of oxides of nitrogen (NOₓ) from fossil fuel combustion contribute to ozone formation via a complex set of photolytic reactions involving hydrocarbons. Most previous modelling effort has been applied to well mixed emissions, at low altitude, in high ozone conditions. In contrast, power station plumes are emitted from high stacks with very little hydrocarbon content. Previous work has shown that ozone concentrations are generally reduced in the plume in the near field, but that at greater distances the plume can have a positive ozone increment. Therefore, any modelling must incorporate considerable complexity in both plume dynamics and chemistry.

Ozone levels may be affected over a regional scale; however, specific local conditions may mean greater attention is needed in the near-field. This has been the case for one of our reference environments, and thus, as with other pollutants we have included greater modelling resolution local to the emission. In addition to regional impacts, the release of power station NOₓ to the free troposphere may affect ozone at a global level. Preliminary models of regional and global ozone formation have been used but we attach a lower level of confidence in the output, and results from these models are currently not used to generate damage costs. Improved models are being developed under various programmes, and it is hoped that these will be available in the near future.
3.3 Considerations on the Appropriate Range of Analysis

When analysing pollutant emissions in studies of this nature, an important question is the range of analysis which should be used. In order to be comprehensive it is essential that impacts are assessed over their full range, with respect to both space and time. To artificially constrain the area of analysis (i.e. by fixing the range of analysis to a model output) will underestimate the full impacts of air pollution. However, there are no clear guidelines for the appropriate system boundaries for impact assessment in terms of geographical distance from the source of pollutants. For the purposes of this work, the guidelines used for traditional environmental assessment of power projects whereby consideration is limited to local impacts, will certainly not apply.

Over the course of the ExternE Project, our policy towards the range over which impacts should be assessed has changed, as our understanding of the damage processes has developed. For some impacts, it was clear that regional analysis was necessary. For example, it would be both scientifically and politically unacceptable to ignore the impacts of long range transport of acid emissions on the lakes of Scandinavia. However, for other pollutants and their impacts, the picture was not as clear. Our initial assessments for health impacts were limited to a distance of tens of kilometres from the source, assuming that the main portion of impacts would occur within this area. To an extent, this limitation was based on the reliability of the Gaussian plume models used for air transport modelling rather than on an analysis of the physical or biological processes leading to damage in the affected receptors. With the development of greater scientific consensus and improved modelling tools we have extended analysis of all pollutants to a regional range. However, where possible, a greater resolution has been used in the local area of the plant. This does mean that we are extending the range of the models that for which they were originally designed. In our view, the errors introduced by using the models outside their design range are likely to be less than the effects of arbitrarily constraining the range of the reference environment.

In order to include as much of the potential damage as possible, it is necessary to know how far the regional range assessments should extend. Relatively simple approximations (European Commission, 1995b) can be used to show the suitable range of analysis. For the purposes of demonstration, the impacts of atmospheric pollutants on human health are used.

A simple calculation can be produced, assuming a homogenous population distribution over Europe. By neglecting the escape of pollutants into the free troposphere and assuming a homogeneous distribution of wind direction during the year, air pollutants are effectively mixed within a cylinder. The top and bottom of the cylinder respectively being the top of the atmospheric mixing layer and the ground surface. Simple dispersion equations can be used to calculate the concentration at distance from the source. These assume average annual wind speed and instantaneous vertical mixing throughout the height of the mixing layer, with values from the Harwell Trajectory Model being used for the mixing layer height, wind speed and dry deposition; chemical transformation and wet deposition of pollutants are not included.

The concentrations can be linked to dose-response functions for health effects to calculate the proportion of damage with distance from the source. Linear functions for health effects with no
threshold of effect are used although it should be stressed that some scepticism is warranted on the extrapolation of the available exposure-response functions down to very small increments far away from the source of pollutants. The resulting graph of the % of cumulative damage with distance is shown in Figure 3.1 for each of the main pollutants (SO$_2$, NO$_x$ and two size fractions of particulate matter).

![Graph showing percentage of cumulative damage expected with distance from the emission source.](image)

**Figure 3.1** Percentage of cumulative damage expected with distance from the emission source.

Although this is only an approximate calculation, the figure clearly shows that limiting the range of analysis to tens of kilometres is insufficient; it only covers a minor fraction of the total expected damage.

Obviously, the basic assumptions (of a homogenous distribution of population, pollutants, conditions and dose-response functions) are not appropriate to estimate the figure of total damage within a selected reference region, and before drawing general guidelines a more detailed analysis of the influence of chemical transformation processes and wet deposition is required. Nevertheless, the results in Figure 3.1 demonstrate that with linear exposure-response functions, a consistent assessment of various impacts within the existing framework should cover distances of several thousands of kilometres from the power plant. We recommend that additional input from area experts is used to give operational guidelines, though for the present analysis, we have extended our range of analysis, where possible, to a European-wide assessment.

The following sections describe the modelling approach we have taken for each of the atmospheric pollutant categories; primary pollutants, acid species and ozone formation. We do not intend to present a comprehensive review of pollutant transport modelling, but some comments on alternative approaches and model limitations are included with the descriptions of the selected models.
3.4 Local Scale Modelling of Primary Pollutants

Close to the power plant, i.e. at distances of 10-50 km, chemical reactions in the atmosphere have little influence on the concentrations of primary pollutants (provided NO and its oxidised counterpart NO$_2$ are summarised as NO$_x$). Due to the emission height from the top of tall power station stacks, the near surface ambient concentrations of these primary pollutants close to the stack depend heavily on the actual vertical exchange of the lower atmosphere. Vertical mixing depends on the atmospheric stability and the existence and height of inversion layers (whether below or above the plume). For these reasons, a model which neglects chemical reactions but is detailed enough in the description of turbulent diffusion and vertical mixing is the most efficient means of assessing ambient air concentrations of primary pollutants on a local scale. Gaussian plume models are commonly used for these estimations of local scale pollutant dispersion from continuous emissions at point sources.

3.4.1 Gaussian plume models

Gaussian plume models assume that the concentration distribution at the surface from a continuous release into the atmosphere has a Gaussian shape, calculated by:

\[ c = \frac{Q}{2\pi u \sigma_y \sigma_z} \exp\left(-\frac{y^2}{2\sigma_y^2}\right) \exp\left(-\frac{h^2}{2\sigma_z^2}\right) \]  

where:
- $c$ = atmospheric concentration;
- $Q$ = the emission rate;
- $u$ = wind speed;
- $y$ = distance from the plume;
- $h$ = stack height;
- $\sigma_y$ = crosswind standard deviation (a measure of plume width);
- $\sigma_z$ = vertical standard deviation.

The assumptions embodied in this type of model usually include those of flat terrain and constant meteorological conditions so that the plume travels with the wind in a straight line at a constant speed. Dynamic features which affect the dispersion, for example vertical wind shear, are ignored. These assumptions generally restrict the range of validity of these models to a region within 50-100 km of the source. The straight line assumption is most strongly justified for a statistical evaluation of long periods (one year in this study), where mutual changes in wind direction cancel each other out, rather than for an evaluation of short episodes.

Additional terms are included to account for reflections from the ground and from an elevated boundary layer inversion which acts as a barrier to vertical transport. The plume rise due to momentum and thermal effects must be estimated to add to the value of $h$. The plume standard deviation as a function of position and weather conditions are important parameters in this model, and much attention has been focused on providing parameterisations to calculate their value from available meteorological data since the early work of Pasquill (1961). A general review of plume models and some parameterisations for $\sigma_y$ and $\sigma_z$ is given in IAEA (1986).
Two types of Gaussian plume model were initially used to estimate dispersion for analysing the UK coal plant. For long-term continuous releases, the annual average concentrations, (equation [1]) can be integrated across wind to remove the dependence on \( \sigma_y \), and a sector analysis done using wind frequency information. Clarke (1979) gives details of the use of this type of model. For concentrations over shorter time periods, equation [1] was used directly. In this case it is important to include the significant variation in \( \sigma_y \) with the length of time scale considered. This variation arises because of the contribution of larger scale eddies to the cross-wind dispersion, usually described as "plume meandering". A procedure described by Moore in Clark (1979) was used to estimate this effect.

The models described above use parameterisations for \( \sigma_z \), the vertical plume standard deviation in terms of distance downwind, the wind speed as measured at the surface, and the solar radiation input. These inputs were thought to be sufficient as descriptions of turbulence induced by mechanical wind shear and by buoyancy forces. Substantial progress has been made in our understanding of turbulent characteristics in the planetary boundary layer (Wyngaard, 1984) since the development of these models. One important result is that it has been found that the turbulent characteristics of a convective boundary-layer are determined largely by the boundary layer depth, \( Z_i \), and a convective velocity scale: \( w^* \) given by:

\[
w^* = \left[ \frac{gQ_oZ_i}{\rho_a C_p T_a} \right]^{1/3}
\]  

where:

- \( g \) = gravitational acceleration;
- \( Q_o \) = the surface heat flux;
- \( \rho_a \) = air density;
- \( C_p \) = specific heat at constant pressure;
- \( T_a \) = air temperature.

Weil and Brower (1984) describe a Gaussian plume model for dispersion from tall stacks based on convective boundary layer scaling. The model predicts that unstable and neutral conditions occur more frequently than in previous models, resulting in higher ground level concentrations. An evaluation of the model using SO2 monitoring data obtained at various distances downwind proved it to be significantly better in performance. This model was also used in the original assessment of UK impacts at West Burton to assess local pollution concentrations for primary emissions of SO2, NOx and particulates.

For the Lauffen plant in Germany, the local range atmospheric transport was calculated using the Industrial Source Complex Short Term model (ISCST2), version 2, of the U.S. EPA (Brode and Wang, 1992). The model calculated hourly concentration values of SO2, NOx and particulates for one year at the centre of each small 10 x 10 km EUROGRID cell, centred on the Lauffen plant. The EUROGRID co-ordinate system is based on geographical parallels and meridians and defines equal-area projection gridcells of 10 000 km\(^2\) and 100 km\(^2\). The width of the large gridcells is defined as a sector of 1.5° on the respective meridian. The small gridcells, used for this local analysis are defined by dividing each side of the large gridcells in ten regular parts. To guarantee high quality data for the local range analysis near the power plant, recent
German data from the ‘Statistik Regional’ database (Statistik Regional, 1993) were transformed to the small EUROGRID gridcells.

The effects of chemical transformation and deposition were neglected. Annual and seasonal mean values were obtained by temporal averaging of the model results. The $\sigma_y$ and $\sigma_z$ diffusion parameters were taken from BMJ (1983). This parameterisation is based on the results of tracer experiments at emission heights of up to 195 m (Nester and Thomas, 1979). More recent mesoscale dispersion experiments confirm the extrapolation of these parameters to distances of more than 10 km (Thomas and Vogt, 1990).

The ISCST2 model provides a simple procedure to account for terrain elevations above the elevation of the stack base:

- The plume axis is assumed to remain at effective plume stabilisation height above mean sea level as it passes over elevated or depressed terrain;
- The effective plume stabilisation height at point $(x,y)$ is given by:

$$h_{stab} = \max(0, z_{stack} - z(x,y) + h_{stack}) + h_{rise}$$

where:

- $h_{stab}$ = is plume stabilisation height;
- $z_{stack}$ = height above mean sea level of stack base;
- $z(x,y)$ = terrain height above mean sea level at point $(x,y)$;
- $h_{stack}$ = stack height;
- $h_{rise}$ = height of plume rise above stack.

and:

- the mixing height is terrain following;
- the wind speed is a function of height above the surface.

Mean terrain heights for each grid cell were calculated from a digital terrain elevation model (DHM, 1990). To investigate the sensitivity of model results with respect to the resolution of terrain elevation, a control run with an enhanced resolution was performed. Results of this control run with 100 x 100 grid cells (each small grid cell divided into 100 even smaller cells) were in good agreement with the results for standard resolution.

The meteorological input data requirements of the ISCST2 model comprise hourly mean values of mixing height, wind speed and wind profile exponent, wind direction at the effective plume height, ambient air temperature and vertical temperature gradient. These data were provided as output of the REWIMET wind field model (Heimann, 1985; VDI-Guideline, 1992). Furthermore, ISCST2 needs hourly stability classes. These were obtained from the REWIMET output on surface roughness and Monin-Obukhov length, according to Golder (1972). REWIMET is initialised by a vertical profile of potential temperature and pressure, and the geostrophic 850 hPa wind each day at 0 UTC. Geostrophic wind and near surface temperature are updated at 3 hour intervals. These data, for surface station 10739 (Stuttgart-Schnarrenberg), are published by the German Weather Service in the European Meteorological Bulletin (DWD, 1990). Geographic input data for REWIMET are terrain elevation and land use. Terrain elevation is used as described above with ISCST2, land use is taken from Statistische Berichte 1991). An example of the results of local scale pollutant transport calculations is shown in Figure 3.2.
Figure 3.2 Local scale incremental TSP concentrations due to the operation of the reference coal fired power plant at Lauffen in the Baden-Württemberg region of Germany.

3.5 Regional Scale Modelling of Primary Pollutants and Acid Deposition

With increasing distance from the power station, emission plumes are spread vertically and horizontally due to atmospheric turbulence. Outside the local area (i.e. at distances beyond 50 km from the stack) it can be generally assumed that the pollutants have been vertically mixed throughout the height of the atmospheric mixing layer. In contrast, chemical transformations and deposition processes can no longer be neglected on this regional scale. The most efficient way to assess annual, regional scale pollution is via models containing a simple representation of transport but a detailed enough representation of chemical reactions.

With the exception of ozone, the main species of interest in the regional assessments are the acidifying pollutants, formed from the primary emissions of SO₂ and NOₓ. A brief description of the processes involved in their regional transportation and the models we have used to assess them are presented in the sections below.

3.5.1 Acid deposition processes

Acid deposition, caused by emissions of SO₂ and NOₓ, has been studied in Western Europe over many years. During this time it has been realised that the acid rain phenomenon involves complex relationships between the emitted precursor species and other atmospheric species, and that there may be difficulty in distinguishing between the environmental effects of acid rain and those from photochemical oxidants such as ozone. Acid rain is a regional scale
phenomenon. The sites where, for example, ecological damage is being experienced are often separated from the sources of pollution by large distances. Because of the complexity between the source and receptor, acid deposition modelling has a major part to play in distinguishing the role of atmospheric chemistry, and models are the only method of practically revealing the source receptor relationships.

The processes involved in modelling acidic deposition include:
- Emission of pollutants;
- Dispersion;
- Atmospheric transport over regional scales;
- Chemical transformations and dry and wet deposition processes.

The principal emissions of interest from fossil-fuel use are SO$_2$ and NO$_x$. These compounds may be removed directly by dry deposition and can also be transformed to other gaseous and particulate phase species and be subsequently removed by dry and wet deposition.

Because of the large geographical range of the acid deposition phenomenon, an extensive emissions database is required. The data for SO$_2$ and NO$_x$ have been taken from EMEP's 150 by 150 km grid (Sandnes and Syve, 1992). However, where modelling has been undertaken on a UK scale only, a more detailed emissions data have been used on a 20 km grid scale (Lee and Johnson, 1993).

Sulphur and NO$_x$ from natural sources (i.e. plankton, soil microbes, lightning, etc.) are important on the global scale (e.g. Spiro et al, 1992; Logan, 1983). The modelling activities described here do not include any natural sources and as more inventories of emissions become available (e.g. Graedel et al, 1995) these can be included in the future to assess their importance on the regional scale.

Smith (1991) discusses the available databases for emissions of SO$_2$, NO$_x$ and NH$_3$ for western Europe. The accuracy of the estimates for European countries was considered to be around 10-20% for NO$_x$ and SO$_2$, and around 50% for NH$_3$. The accuracy tends to decrease with increased resolution. The uncertainties in emissions may therefore make a considerable contribution to the final uncertainty of model results.

Meteorological processes are important to several aspects of acid deposition. These include initial dispersion and long range transport processes, reviewed by Eliassen (1980; 1984), Johnson (1983) and Schwartz (1989), together with those processes involved in the formation of cloud droplets and rainfall. These processes are important because of the role of aqueous phase chemistry in the oxidation of SO$_2$ to sulphate.

It is important to consider the nature of particulate species predicted by such modelling. Only secondary species are included and other particulates from fossil-fuel combustion, e.g. fly ash from power stations or carbonaceous material from the transport sector, are not considered. In addition, larger particles, i.e. 1-10 µm, may arise from sources such as soil erosion, sea-spray, open mining etc. Again, these are not treated by the models described here.
3.5.2 Types of acid deposition models

Several different types of model have been used to investigate acid deposition. These include Eulerian grid models, Lagrangian trajectory models and statistical models. These have been discussed in detail by several authors (Johnson, 1983; Eliassen, 1980, 1984; Hough and Eggleton, 1986; Schwartz, 1989). The principle types of model are discussed here, together with a more detailed description of the model used in this study.

In the Eulerian approach the model domain is mapped onto a grid, with the emissions to each grid cell specified. Atmospheric transport between cells is incorporated, together with chemical processes. The complexity of the time dependent meteorological processes and emissions required as input into this type of model may restrict the grid size to be large, and subgrid scale mixing in the model makes it inappropriate for studying changes to individual stack emissions. Despite these problems, much effort is being directed into this type of model because it can address problems of non-linear chemistry and also the combined problem of tropospheric oxidant production together with acid deposition. It is costly to obtain climatological estimates of acid deposition from this type of model because of the long time required for model runs.

Lagrangian models consider air parcels which move with the direction and velocity of the wind. Eliassen (1984) provides a review of some aspects of Lagrangian models of air pollution. There are two main types of these models; those orientated towards the source of pollution and those which are receptor-orientated. In the first case, the source provides an initial mass of pollutant to the model air parcel which subsequently moves away from the emission site. In the receptor-orientated case, the air parcel moves over various emission sources until it arrives at the receptor site. Lagrangian models permit the inclusion of more detailed chemistry than the Eulerian schemes, but the role of mixing between air parcels with different origins is not included. The effects of wind shear, which give different trajectory paths to parcels of air in different levels in the atmosphere, is seldom considered as the common assumption is that most of the pollution is confined to the mixing layer. Nevertheless, despite these theoretical problems, Lagrangian models have proved useful because their sensitivity to individual emission contributions can be rapidly assessed. Indeed, Lagrangian type models have proved capable of reproducing the distribution pattern and magnitude of regional sulphate deposition (Schwartz, 1989).

To estimate the regional scale concentration and deposition of acid species, the ExternE Project has used the Harwell Trajectory Model (and derivations). The model was originally described by Derwent and Nodop (1986) for atmospheric nitrogen species, and extended to include sulphur species by Derwent, Dollard and Metcalfe (1988). It is a receptor-orientated Lagrangian plume model employing an air parcel with a constant mixing height of 800 m, moving with a representative wind speed of 7.5 ms$^{-1}$. The results were obtained at each receptor point by considering the arrival of 24 trajectories weighted by the frequency of the wind in each 15$^\circ$ sector. The trajectory paths were assumed to be along straight lines and were started at 96 hours from the receptor point, equivalent to a distance of 2,592 km.
The chemical interconversions and removal mechanisms specified in the HTM are shown in Figure 3.3. These schemes are highly simplified treatments of atmospheric chemistry. The SO$_2$ oxidation scheme has a rate of 1% hr$^{-1}$; such an approach is commonly adopted in modelling sulphur oxidation (Binkowski et al., 1990) as detailed modelling of heterogeneous chemistry is not straightforward. The chemical scheme for oxidised nitrogen species assumes an averaging of day- and night-time chemistry. In general, dry deposition of gases is important close to sources and wet deposition more important at distance from sources. Dry deposition of particulate species is more evenly spread as the deposition velocity of material in the 0.1-1 µm size range is generally quite small.

![Chemical reactions of the sulphur and nitrogen species included in the Harwell Trajectory Model.](image)

**Figure 3.3** Chemical reactions of the sulphur and nitrogen species included in the Harwell Trajectory Model.

Unusually, the HTM also has a scheme for HCl. Hydrochloric acid is removed quite quickly (by wet and dry processes) close to sources and is much less important in terms of acidity than SO$_2$ and NO$_x$. It should also be noted that in terms of ecosystem acidification, NH$_3$ makes an important contribution because deposited ($\text{NH}_4$)$_2\text{SO}_4$ releases two protons as a result of microbiological processes in the soil. Ammonia is also an important chemical 'regulator' as
oxidation of $\text{SO}_2$ in clouds is pH dependent and thus, can be self-limiting. However, the presence of $\text{NH}_3$ raises the pH and allows more $\text{SO}_2$ to be converted to $\text{H}_2\text{SO}_4$ (Behra et al., 1989). Furthermore, $\text{NH}_3$ plays an important role in oxidised nitrogen chemistry forming $\text{NH}_4\text{NO}_3$ particulate.

The HTM has more recently been updated (Lee and Johnson, 1993), using UK emissions on a 20 km grid scale and most significantly, a rainfall field on a 20 km grid which substantially improves the model's ability to reproduce observations over that reported by RGAR (1990). For the UK implementation, the regional range is limited to the national level, with data output on the UK National Grid. An example of the model output is shown in Figure 3.4.

For the analysis of coal and gas fired power plants in the UK, use of the Harwell Trajectory Model output was restricted to the UK National boundaries. Transport beyond the UK was determined using the output of the European Monitoring and Evaluation Programme (EMEP). This uses a complex and well reviewed modelling procedure to calculate the deposition of sulphur and nitrogen in all European countries as a function of the source country. The most recent results include data up to 1991 (EMEP, 1992). This source-receptor national matrix was used to calculate the fraction of each country's emissions deposited in each other country for both sulphur and oxidised nitrogen species. It is then assumed that this fraction can be applied to the output of an individual power station, assuming that the resulting pollution field is uniform over the whole of the receptor country. At the ranges typical of transboundary pollution these are reasonable approximations. The largest source of error is likely to derive from neglecting the effects of differential precipitation on wet deposition.

Subsequently, for the modelling of the coal, lignite and oil plants in Germany, a version of the Harwell model was adapted at IER. This is called the Windrose Trajectory Model and is designed to operate on the EUROGRID co-ordinate system. Primary and secondary acid species were modelled across Europe. The spatial resolution for this regional analysis is output on the large EUROGRID cells (100 x 100 km). The Windrose model also considers receptor specific meteorological input data. Where available, 1990 meteorological data were taken from the Meteorological Synthesising Centre-West of EMEP at The Norwegian Meteorological Institute (Hollingsworth, 1987; Nordeng, 1986). 6-hourly data in the EMEP 150 km grid of precipitation and wind (at the 925 hPa level) were transformed to the EUROGRID grid and averaged to obtain the receptor-specific mean annual windrose (frequency distribution of the wind per sector), mean annual windspeed and total annual precipitation. Base line emissions of $\text{NO}_x$, $\text{SO}_2$ and $\text{NH}_3$ for Europe were taken from the 1990 EMEP inventory (Sandnes and Styve, 1992). An example of model output is shown in Figure 3.5.
Figure 3.4  
Harwell Trajectory Model output (incremental ground level NO\(_x\) (ppb) increase) from the UK reference gas fired power plant at West Burton.
Figure 3.5 Regional scale incremental $\text{SO}_2$ concentrations due to the operation of the reference combined cycle oil fired power plant at Lauffen.
In addition, primary particulate emissions from the power plant were also modelled across the same co-ordinate system using the simple wind trajectories within the Windrose Trajectory Model. The issue of system boundaries was investigated using the model; the results showed 95% of the emitted particulate matter was deposited within the model domain. According to Figure 3.1, this percentage corresponds to an effective downwind distance of some 1,900 km for TSP (5-10 mm). The actual boundaries of the model domain are 1,600-2,400 km from the plant. As the Windrose model takes account of both dry and wet deposition and tends to over-estimate sulphate in precipitation, the total deposited sulphur (92% of the emitted mass) and nitrogen (78%) are somewhat higher than the values in Figure 3.1 at a distance of 1,900 km from the source.

3.6 Ozone Modelling

The assessment of ozone formation generated by fossil fuel power station emissions has proved to be one of the most difficult modelling exercises within the ExternE Project. Ozone formation is critically influenced by specific weather conditions and requires large quantities of certain precursors, which frequently arise from other trans-boundary pollution sources.

In general, there are sufficiently large scale emissions of precursors over large parts of Europe, in magnitude and composition, to form regional scale ozone episodes. The occurrence of photochemical episodes therefore depends largely on meteorological conditions; for regional scale ozone formation, the most important (PORG, 1993) are:
- Sunshine to drive the chemical reactions;
- Low wind speeds (0-5 m/s) to inhibit dispersion;
- Restricted boundary layer depths, to allow the build-up of precursors;
- High air temperatures (above 20°C) to enhance evaporative emissions of hydrocarbon precursors and promote certain chemical reactions.

The meteorological conditions which fulfil all these requirements are those associated with stable summer anticyclones (high pressure weather systems). The specific processes which contribute to surface ozone ($O_3$) concentrations may be classified (Altshuller, 1986) as:
- Transport of stratospheric $O_3$ towards the surface;
- Photochemical $O_3$ formation within the free troposphere and unpolluted parts of the planetary boundary layer;
- Photochemical formation within the polluted boundary layer;
- $O_3$ formation within plumes from single or multiple emission sources.

The major routes for the chemical formation of ozone are processes which convert NO to $NO_2$, which is subsequently photolysed to form NO and an oxygen atom which combines with oxygen to form ozone:

\[
RO_2 + NO \rightarrow NO_2 + RO \quad [4]
\]
\[
NO_2 + hv \rightarrow O + NO \quad [5]
\]
\[ O + O_2 \xrightarrow{M} O_3 \] \[6\]

where RO₂ is HO₂ or an organic peroxy radical formed from the oxidation of hydrocarbons, \( h\nu \) is radiation (280< \( \lambda < 430 \) nm) and M is any molecule such as N₂ or O₂ which acts to dissipate the energy released by the reaction in order to prevent O₃ decomposing. Ozone is also destroyed by photochemical processes, including the reaction:

\[ NO + O_3 \rightarrow NO_2 + O_2 \] \[7\]

For occasions when equation [7] is the dominant loss process, equations [5], [6] and [7] can be used to establish an equilibrium relationship between O₃, NO and NO₂:

\[ [O_3] = \frac{[NO_2]}{[NO]} \cdot \frac{k_2}{k_4} \] \[8\]

where \( k_2 \) is the rate of photolysis of NO₂, and \( k_4 \) is the rate of the reaction between NO and O₃ (equation [7]). Thus, where the NO₂:NO ratio is low, as in the initial stages of a power station plume, the ozone concentration remains low.

The spatial distribution of the man-made photochemical oxidant precursor emissions needed for ozone generation tend to be concentrated in the urban and industrial areas of Northwest Europe. During these episodes the concentration rises to over 100 ppb - 2 to 3 times the background concentration.

Therefore, reference power station emissions are likely to affect regional ozone formation. However, the nature of plume emissions are very different from most ozone modelling exercises to date, namely the effects of ozone for stable atmospheric conditions in highly polluted areas. In these cases, the major culprits are vehicle emissions, which contain both NOₓ and hydrocarbons, and therefore, most modelling effort in the past has been applied to well mixed emissions, at low altitude, in high ozone conditions. In contrast, emissions emitted from high stacks contain very little hydrocarbon content and initially have high concentrations of NO which break down ozone.

### 3.6.1 Models of regional ozone formation

Regional air quality models for ozone have been developed in order to understand the relationship between peak ozone concentrations and emissions. Several types of model have been used, and Seinfeld (1988) provides a critical review. The models are divided broadly into those which use a fixed Eulerian grid approach within a specified area, and those which use a Lagrangian technique where an air parcel is followed over a specified trajectory. Hybrid models incorporate both methods within the same model. Within this broad classification, many techniques have been used to specify meteorological features used in models and many different chemical mechanisms of varying complexity have been used.
Results from both measurements and box model studies have shown that ozone production per unit of NO\textsubscript{x} lost (the ozone production efficiency) varies non-linearly with the NO\textsubscript{x} concentration (Liu \textit{et al}, 1987; Lin \textit{et al}, 1988). Significant differences are found in ozone production efficiency between models with different non-methane hydrocarbon (NMHC) concentrations and compositions. These results have important implications to the modelling of ozone production from discrete sources such as power plant plumes as distinct from sources such as traffic emissions which have a wide distribution. Another difference is that power plant plumes are characterised by high NO\textsubscript{x} to NMHC ratios compared to emissions at the surface, where the two are often released together.

Eulerian or Lagrangian box models which have a large grid size tend to smooth the variability in the NO\textsubscript{x} concentration because the emission sources are artificially mixed in a large volume of air. The amount of ozone produced in the model is therefore higher with models of large grid size, compared with models which resolve the NO\textsubscript{x} emissions distribution. Sillman \textit{et al} (1990) demonstrate increases in peak ozone concentrations with models of grid sizes of 80 x 80 km\textsuperscript{2}. In contrast, Seinfeld (1988) quotes model results from Urban Airshed models where peak O\textsubscript{3} was reduced considerably by an increase in grid size. It is clearly necessary to simulate to some extent the frequency distribution of NO\textsubscript{x} in the model.

The lifetime of ozone concentrations in the boundary layer implies that a regional scale simulation is required. This makes it computationally difficult to use fine-mesh models. The scale of the model domain used must be related to the lifetime of the important model precursors for ozone. In the case of NO\textsubscript{x}, lifetime is largely determined by conversion to HNO\textsubscript{3} through the reaction of NO\textsubscript{2} with OH. As OH concentrations are often reduced initially within power plant plumes (Cocks and Fletcher, 1988, 1989) the NO\textsubscript{x} lifetime will not be reproduced adequately by a Eulerian Grid model with large grid sizes, and the time and length scale of the model will need to be adjusted accordingly. Sillman \textit{et al} (1990) give chemical loss lifetimes for a power plant plume several times greater than for Eulerian grid models.

3.6.2 Measurement and modelling of ozone in power station plumes

The role of NO\textsubscript{x} emissions from power stations on local air quality depends on the rate of dispersion of the plume and on the chemistry within the expanding plume. Where rapid mixing of the plume into the ambient air occurs, ozone formation is expected. More typically, the plume remains intact and aloft for a considerable distance downwind from the point of emission and can thus make no contribution to ozone concentrations at the surface. In certain conditions the plume remains intact below a stable layer and can be recognised at hundreds of km from the source. Therefore, models which do not include plume dispersion and chemical processes will overestimate the effect of NO\textsubscript{x} emitted from tall stacks on ozone concentrations, especially close to the emission point. This is because the emitted nitrogen oxides are assumed to be mixed into a much larger volume of air than is actually the case, and greater concentrations of non-methane hydrocarbons are present than in a plume. The prediction of changes in the surface concentration of ozone in response to power plant NO\textsubscript{x} emissions remains a major problem because of the uncertainties involved in modelling plume dispersion and mixing processes.
The uncritical application of Eulerian grid or single box models in order to calculate the effects of high stack NO$_x$ emissions on surface ozone concentrations may give erroneous results for several reasons, including:

- The assumption of instantaneous mixing within grid cells, creates errors in calculating the mean NO$_x$ concentration;
- The conclusion from reactive plume models in European conditions is that ozone excesses are not predicted until after many hours of travel;
- The vertical resolution of many Eulerian grid models is insufficient and may lead to NO$_x$ concentrations aloft from stack releases mixing too rapidly with hydrocarbons released at the surface;
- The NO$_x$ lifetime in the plume is expected to be greater than NO$_x$ in the ambient air mass.

Within the plume, the reaction of NO with O$_3$ present in the ambient air tends to reduce O$_3$ concentrations (see equation [6]). This reaction, together with the rate of photolysis of NO$_2$ determines a photo-stationary state which may be attained in the plume. Oxidation of NO is limited by the amount of oxidants present in the expanding plume, and the NO:NO$_2$ ratio will remain high. Conversion of NO$_x$ to HNO$_3$ within the plume may be an important removal process, and be sufficiently rapid to act before the NO$_x$ is widely dispersed.

Measurements of O$_3$ in power plant plumes confirm the decrease in O$_3$ in the initial stages of dispersion. For example, Altshuller (1986) summarises measurements in America that suggest that several hours of transport are necessary before ozone will be produced in the plume. In some circumstances ozone deficits are still present after over 500 km of travel (Clark et al., 1984). Models of the chemical reactions and dispersion processes affecting power station plumes have been constructed (Miller et al., 1978; Hov and Isaksen, 1981; Stewart and Liu, 1981; Derwent, 1982; Cocks and Fletcher, 1988, 1989; Janssen, Nieuwstadt and Donze, 1990). The major features of these models are similar in that an initial decrease in O$_3$ is followed by an increase after several hours of travel.

Plume models have been used to calculate O$_3$ concentrations in European conditions (Derwent, 1982; Cocks and Fletcher, 1988; 1989). Derwent (1982) used a vertically-integrated plume model with 20 horizontal cells to study the development of chemical species extending from a large isolated UK power station downwind over areas of differing background emissions. The plume was studied for 10 hours of daytime travel. In all cases there was a significant ozone decrease in the plume at all times. In one case a very small increase above ambient concentrations was observed at the edge of the plume. The author concluded that power station plumes have little significant impact on photochemical ozone formation in rural regions of the UK. Cocks and Fletcher (1988, 1989) used a single box model to study the advection of a 2,000 MW power plant plume over a 24 hour period. Studies were made with the plume dispersing into typical 'rural' ambient air, and into an 'urban' ambient air, the latter with much higher emissions of NO$_x$, CO and hydrocarbons. In both cases ozone concentrations in the plume were normally predicted to be lower than in the ambient air mass, and significant ozone excess was only predicted for slowly dispersing plumes in summer conditions after 24 hours. Most of the plume NO$_x$ was predicted to be converted into HNO$_3$ after this time. From the results discussed above, it appears that ozone may not be generated until some 100 km or more from the point of emission.
However, studies with plume models in the conditions of USA (Hov and Isaksen, 1981; Stewart and Liu, 1981) have shown ozone increases above ambient air within a few hours of travel. Derwent (1982) considers this almost certainly due to the differing hydrocarbon to NOx ratios in the emissions. Seinfeld (1988) expressed the need for a grid based photochemical model with a subgrid scale approach to plume dispersion and chemistry. An example of this type of model is described by Sillman et al (1990) and applied to an area of the continental USA.

Similarly, there will be some European examples, where in the presence of conditions which are highly favourable to O3 generation, ozone concentrations may be increased within a relatively short distance of power plants. This has been the case for one of the reference locations chosen - at Lauffen in Germany. A highly detailed ozone model has been used for predicting the local ozone concentrations from fossil fuel power station emissions for this area. It is important to stress that there are several major VOC emitters located in the neighbourhood of the power plant, leading to highly site specific effects on ozone formation. However, because of the complexity of the model, the results generated have been restricted to a local zone of only about 150 km by 150 km; the effect of emissions on regional ozone have not been calculated.

For this specific local analysis, changes in the surface concentration of ozone were simulated by Fiedler et al, at IMK (Institut für Meteorologie und Klimatforschung) at the University of Karlsruhe. They employed the mesoscale Eulerian model KAMM (Karlsruhe Atmospheric Mesoscale Model) to investigate peak ozone concentrations during a summer day. The KAMM model system comprises a non-hydrostatic meteorological driver (Adrian and Fiedler, 1991) and a Eulerian transport and chemistry model (Nester and Fiedler, 1992), the latter employing the gas phase mechanism of RADM-2 (Regional Acid Deposition Model) (Stockwell et al, 1990).

The model grid was divided into 57 x 53 x 18 cells, representing 171 km x 159 km x 4600 m. The area is defined by 58 x 54 gridcells with a size of 3 km x 3 km on the German Gauß-Krüger co-ordinate system. The depth of the surface layer was 24 m, the depth of the elevated layer at plume height was 220 m. A base line scenario was available from a prior study (Vogel et al., 1994), which was based on meteorological and emission data of a photosmog episode on 3 August 1990, when half-hour ozone concentrations as high as 208 µg/m$^3$ were observed. A model run with additional emissions from the reference power plant was compared with the base line study to obtain incremental concentrations. Figure 3.6 shows these incremental concentrations, in the surface layer of the model, at 2 pm, the time of highest ozone levels. It has to be emphasised, that due to the complexity of the KAMM-DRAIS model, simulation of ozone formation has been done only for a single reference day during a summer ozone episode.
Greater problems have been encountered with regional modelling. There is a considerable problem in calculating the amount of O$_3$ formed from high-stack emissions during the range of conditions found over a year. Models of plume chemistry assume that NO$_x$ is contained within the boundary layer, and that clear sky conditions prevail. These conditions are not often met in practice, though they offer ideal conditions for O$_3$ generation. In cloudy conditions, and where vertical air motion is not restricted by a boundary layer inversion, NO$_x$ may be carried into the free troposphere and generate ozone. This may then be sufficiently long lived to affect ozone concentrations over extensive areas of the globe.

In view of the problems of calculating O$_3$ concentrations in the varying conditions over a period as long as a growing season, two illustrative cases were considered for the UK implementation. In the first case, conditions were taken to be relevant to a summer anticyclonic circulation. In the second, conditions of free convective transport into the troposphere were assumed to apply. The first case (regional modelling) was addressed using a vertically averaged plume model of the type described by Derwent (1982); the second using a zonally averaged global model of tropospheric chemistry (Hough, 1989, 1991).

### 3.6.3 Regional scale ozone modelling

The model used for the regional summer anticyclonic assessment is based on Lagrangian photochemical plume modelling techniques (Derwent and Hov, 1979; Derwent 1982; Stewart and Liu, 1981). The boundary layer is modelled by following a moving parcel of air downwind over a straight line trajectory, where the upper boundary of the moving box and the
wind speed are specified as a function of time of day. Plume behaviour is modelled using twenty vertically averaged cells across the plume width. These cells expand with the growth of the plume, and there is an interchange of material between the cells and between the boundary layer and outer plume elements. Only the turbulent component of plume spreading is considered, as plume meandering processes which occur through variation in wind direction are unimportant in establishing chemical concentrations in the 'instantaneous' plume. Jones (1981) describes the plume spreading angle for long range atmospheric transport to be:

$$\theta = x^{-0.16} \tag{9}$$

where:

- $x = \text{distance (m)}$.

To assess the importance of the results from this type of model, a scenario was considered where nocturnal emissions from a power station were followed over a trajectory for two days. The concentrations predicted by the model are shown in Figure 3.7, and the average differences from the air outside the plume in Figure 3.8. As the figures show, initially the plume ozone concentrations are considerably reduced in comparison to those in the boundary layer, because of the reaction of NO with $O_3$ in the plume. Only during the second day do $O_3$ concentrations within the plume exceed those outside it. These values were extrapolated to the UK, assuming an equal probability of wind direction. The results were multiplied by the frequency of the wind in each sector, and by the proportion of the sector occupied by the plume. Annual values were then derived by assuming the frequency of anticyclonic conditions in each month to provide average ozone increments for the UK.

### 3.6.4 Global ozone modelling

For the global ozone formation modelling, the Harwell two dimensional model of the global troposphere was used to estimate $O_3$ increases due to additional NO$_x$ emissions escaping into the free troposphere. The model used is described by Hough (1989, 1991) and consists of a description of transport and chemical processes in the domain from the surface to 24 km and from pole to pole. Table 3.2 shows the ozone increments from an additional NO$_x$ input. These results illustrate the seasonal variability of ozone production from NO$_x$ sources. Under winter conditions photochemistry is weak and NO$_x$ acts to reduce ozone in the latitude of release. The largest ozone increments are in July.

However, because of uncertainties in implementing both regional and global ozone models, we have not used the output from these models to quantify impacts in the present project. The location of the reference coal and gas plants in the UK, i.e. close to the east coast, and the nature of power station plumes, means the model must be extended over much of Europe to include most of the positive ozone increment. Because of problems in extending to this range, we have not quantified the impacts of the output. Similarly, the formation of ozone on a global scale are difficult to translate into impacts.
**Figure 3.7** Average ozone concentrations predicted by the plume model.

**Figure 3.8** The difference between the average plume ozone concentration predicted by the plume model and those in the boundary layer.
Table 3.2  Results of the global ozone model for the Northern hemisphere, showing increased surface level $O_3$ (ppb) resulting from emission of 1 Tg [N]/yr NOx emission in the 52°N cell.

<table>
<thead>
<tr>
<th>Latitude range</th>
<th>January</th>
<th>April</th>
<th>July</th>
<th>October</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.0 - 4.8</td>
<td>0.013</td>
<td>0.002</td>
<td>0.000</td>
<td>0.000</td>
</tr>
<tr>
<td>4.8 - 9.6</td>
<td>0.023</td>
<td>0.006</td>
<td>0.001</td>
<td>0.001</td>
</tr>
<tr>
<td>9.6 - 14.5</td>
<td>0.036</td>
<td>0.014</td>
<td>0.005</td>
<td>0.006</td>
</tr>
<tr>
<td>14.5 - 19.5</td>
<td>0.055</td>
<td>0.030</td>
<td>0.010</td>
<td>0.012</td>
</tr>
<tr>
<td>19.5 - 24.6</td>
<td>0.081</td>
<td>0.056</td>
<td>0.021</td>
<td>0.025</td>
</tr>
<tr>
<td>24.6 - 30.0</td>
<td>0.111</td>
<td>0.097</td>
<td>0.038</td>
<td>0.044</td>
</tr>
<tr>
<td>30.0 - 35.7</td>
<td>0.149</td>
<td>0.158</td>
<td>0.073</td>
<td>0.079</td>
</tr>
<tr>
<td>35.7 - 41.8</td>
<td>0.109</td>
<td>0.264</td>
<td>0.152</td>
<td>0.139</td>
</tr>
<tr>
<td>41.8 - 48.6</td>
<td>-0.110</td>
<td>0.433</td>
<td>0.364</td>
<td>0.200</td>
</tr>
<tr>
<td>48.6 - 56.4</td>
<td>-0.266</td>
<td>0.480</td>
<td>0.917</td>
<td>-0.008</td>
</tr>
<tr>
<td>56.4 - 66.4</td>
<td>-0.194</td>
<td>0.479</td>
<td>0.409</td>
<td>0.089</td>
</tr>
<tr>
<td>66.4 - 90.0</td>
<td>-0.151</td>
<td>0.391</td>
<td>0.274</td>
<td>0.117</td>
</tr>
</tbody>
</table>

3.7 Conclusions

The modelling of the environmental impacts of fossil fuel fired power stations requires knowledge of the dispersion of pollutants from the source and the prediction of formation of secondary pollutants (e.g. $O_3$ and sulphate) from emissions. Similarly, for the assessment of radionuclide emissions from the nuclear fuel cycle, models must be able to include dispersion and basic chemical transformation. The type of model used in any analysis depends on the pollutant in question, and the range of analysis.

The ExternE Project is the first externalities assessment to use high quality atmospheric models over a wide range of spatial scales and pollutants. This represents a significant step forward in the state of the art. However, atmospheric modelling is a specialised and resource intensive exercise, and the appropriate level of approximation in externalities work requires further attention.

Average annual concentrations and depositions of primary and acidifying secondary pollutants were assessed with two simple models. For the local scale analysis, Gaussian plume models were used. These models neglect chemical transformation but allow for some detail with respect to turbulent diffusion of the plume close to the source. They can be reliably used for short range transport of primary pollutants, up to distances of between 50 and 100 km. However, at longer ranges, the procedure is less reliable. Therefore, for the regional scale analysis, Lagrangian trajectory models were used. These employ a chemical mechanism for the simulation of the formation of acidic compounds from the emitted species. The range for this regional assessment has been extended to cover the whole of Europe. Both the above models assume straight line trajectories. These are only appropriate for very short distances (less than 10 km) or for long term averages. In the present report the latter condition applies, since annual averages are estimated. Results of the models used to estimate increases in acidic deposition caused by the incremental power stations are in reasonable agreement with observations and other comparable models.
The prediction of O$_3$ formation from pollutants emitted from high level stacks requires consideration of within-plume processes over considerable distances downwind from the source. The assessment of photochemical oxidants continues to remain a problem within the ExternE Project, though significant advances have been made during the course of the programme. The effort for reliable photochemical modelling is still too high and modelling capacities are still too restricted to allow time- and cost-efficient calculations.

Assessments of worst case increments on ozone concentrations were performed from an application of the model KAMM, these data only extend to the local area, over short time periods. Substantial progress in model development, model evaluation, and compilation of consistent data sets (comprehensive meteorological data, base line emissions and background concentrations) is a prerequisite for a more thorough assessment of photochemical pollution from elevated point sources. It will be a priority task for future studies to adopt appropriate methods when available.

There still remains a large degree of error in the regional modelling of ozone formation because the rate at which the plume mixes with ambient air cannot be accurately predicted, and because the concentrations of hydrocarbons and other species are important. The results of the O$_3$ model described in this Chapter are in broad agreement with observations made on power station plumes. A preliminary assessment of global ozone has also been made though a lower degree of confidence is attached to current models.

Finally, we have identified the most important requirements for future research, these are:

- Development and use of improved ozone formation models. Ozone impacts cannot yet be calculated over a significant distance due to the absence of suitable models in European conditions. Models need to be developed to describe ozone formation due to power station emissions, both in NO$_x$ rich plumes in the boundary layer and in the free troposphere;
- Improved models for regional concentrations of primary and secondary pollutants. The type of approach required for the European-scale modelling requires the more sophisticated approach that the UK implementation of HTM takes, using more detailed information on meteorology and removal process (Lee et al., 1995). Furthermore, account needs to be taken of more extreme topography, e.g. the Alps.
- Improved resolution of the acid rain model in the vertical domain would substantially improve its ability to reproduce surface-level concentrations of pollutants such as NO$_x$ and NH$_3$, and to a lesser extent secondary particulates. The chemistry module could be altered to account for seasonal effects.
- The present incompatibility of grid systems is problematic. The EMEP framework (150 by 150 km) on which emissions are compiled is convenient but provides poor spatial resolution - this has been shown to be inadequate for the UK and the FRG. However, future emission inventories from EMEP will be released on a 50 km basis and conversion algorithms to enable computation and conversion between the grid systems have now been independently developed. Where possible ‘look-up tables’ should be developed to describe pollution transfers between grid cells to minimise the need to re-run models and allow easier dissemination of the methodology.
3.8 References


4. PUBLIC HEALTH EFFECTS OF AIR POLLUTION ARISING FROM FOSSIL FUEL COMBUSTION

4.1 Introduction

4.1.1 The impact pathway

Combustion processes cause an increase in the concentration of certain atmospheric pollutants. Several of these have been associated with adverse health effects within the general public. The magnitude of impacts is estimated following the damage function approach. For the purpose of modelling, the impact pathway representing a set of complex processes is broken down into the following levels (Figure 4.1):

- Emission;
- Transport and chemical conversion;
- Exposure;
- Biological startpoints (biological and physical impacts);
- Physical endpoints (health effects);
- Economic valuation.

Although Figure 4.1 indicates some early markers of response to inhaled pollutants, there is at present no agreed understanding of the mechanisms whereby small changes in ambient air pollution may lead to increased mortality or severe morbidity (e.g. hospital usage) on the same day or soon after. In general, this seems likely to occur only against a background of severe pre-existing ill-health. The precise mechanisms are however unknown; and so Figure 4.1 is necessarily incomplete.

4.1.2 Exposure assessment

Atmospheric modelling suggests that there may be some (though immeasurably small) increase in pollutant concentration even at several hundreds of kilometres from a power plant or other combustion site. The population potentially exposed to these very small increments is therefore many hundreds of millions of people, according to the location of the plant, the dispersion of primary pollutants and formation of secondary pollutants, and population density. Exposure to a pollutant is defined as a person's contact with a pollutant of a certain concentration during a certain period of time (COST, 1992). This can occur through inhalation, ingestion and/or penetration of the skin surface. Taking into account the specific pollutants emitted from a coal fired power plant, inhalation is the most important route for the present analysis and is the only one considered here.
Public Health Effects

Exposure

Mechanism of Effects
- irritant effects on the respiratory system
- increased airway resistance
- biochemical changes
- morphological changes
- increased susceptibility to infections

Mortality / Morbidity
- chronic bronchitis
- asthma
- cough
- ...

Valuation
- Value of Statistical Life
- Value of Restricted Activity
- Value of Asthma Day

Figure 4.1 The Air Pollution - Human Health Impact Pathway.
The air pollutants examined in this Chapter are particulates (both primary and secondary), \( \text{SO}_2, \text{NO}_x \) and ozone. In general, epidemiological studies of the public health effects of these pollutants do not consider personal exposures of the study group. Rather, they examine relationships between health effects and ambient concentrations of pollutants. Implementation in the present project is therefore similarly in terms of changes in ambient concentrations rather than in personal exposures. This simplifies implementation; but has a disadvantage. There is limited knowledge of how changes in ambient air pollution are related to changes in the various indoor environments where many people typically spend most of their time. A problem of transferability of exposure-response functions arises when the relationship between ambient and personal exposures differs between the epidemiological study populations and the target populations at risk in the present project. Such differences can arise, for example, because of differences in human activity patterns or in housing (especially through the use of air conditioning).

Epidemiological studies use a wide variety of measures of particulate matter (PM) ambient pollution. We have used approximate conversion factors between indices as given by Dockery and Pope (1994).

Air pollution levels show substantial temporal and spatial variation. Concentrations are integrated over time to give average values (COST, 1992). In epidemiological studies of acute effects on health, the time period of interest is usually 24 hours, though sometimes multiple day averages are used. Ozone measures typically refer to maximum 1-, 5- or 8-hour

Table 4.1 Reference data and model requirements for the pathway implementation.

<table>
<thead>
<tr>
<th>Stage of pathway</th>
<th>Reference data required</th>
<th>Type of models or relationships</th>
<th>Output units</th>
</tr>
</thead>
<tbody>
<tr>
<td>Emissions</td>
<td>1. Fuel type and composition</td>
<td>1. Emission factors</td>
<td>([g/MWh_{el}])</td>
</tr>
<tr>
<td></td>
<td>2. Amount of fuel used and electricity generated</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>3. Generation technology</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Transport and chemical conversion</td>
<td>1. Existing ambient air concentration of ( \text{SO}_2, \text{NO}_x, ) particulates, VOC</td>
<td>1. Industrial Source Complex Model (US-EPA) (primary pollutants - local scale)</td>
<td>([\mu g/m^3])</td>
</tr>
<tr>
<td></td>
<td>2. Meteorological data: wind speed, wind direction, air temperature, stability class, mixing height, etc.</td>
<td>2. Harwell Trajectory Model (sulphates/acid deposition - regional scale)</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>3. KAMM-DRAIS model (ozone - local scale)</td>
<td></td>
</tr>
<tr>
<td>Exposure</td>
<td>1. Ambient air concentration of pollutants</td>
<td>[ E_i = \sum_{j} C_j \cdot t_j ]</td>
<td>([\mu g/m^3 \times [a]])</td>
</tr>
<tr>
<td>Biological startpoints</td>
<td>1. Population at risk</td>
<td>1. Exposure-response functions</td>
<td>Morbidity and mortality indicators</td>
</tr>
<tr>
<td>Physical endpoints</td>
<td>2. Distribution of risk groups</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
concentrations within a day, rather than to the full 24-hour average. In studies of chronic health effects, the averages refer to time-periods of one or several years. The same timescales are used for implementation in the present study. However, estimated daily acute effects are accumulated over a year and expressed as the effect of changes to annual average concentrations.

By focusing on average concentrations within a given period, the epidemiological studies give no direct information about the relevance, if any, for health of short-term peak concentrations within those periods, though clearly the ozone measures are designed to take some account of within-day variations. Insofar as within-day peaks may be relevant to acute health effects, the similarity of the peak-to-average ratio is one factor influencing the transferability of exposure-response relationships.

Ambient concentrations are typically measured using fixed-point monitors. Large scale mortality or hospital usage studies may include several million people residing in a region covered by several monitoring sites. Usually, a regional average concentration is used, following examination of the similarity of concentrations across sites. No attempt is made to account for time spent outside the study region; the effect of such a refinement would be small.

Similarly with implementation: the vast geographical area to be considered is sub-divided into smaller regions using a regular grid system. Each gridcell is considered as a micro-environment with a homogeneous pollutant concentration. The associated population at risk is the population resident in that gridcell. As in the epidemiological studies, daily movements of people between gridcells are ignored. Effects are estimated separately by gridcell and then accumulated over gridcells.

4.1.3 The ‘new’ air pollution epidemiology

It has long been established from studies of pollution episodes, for example that in the Meuse valley, the London smog episodes of the 1950s and that in Donora Valley, that very high levels of ambient air pollution are associated with increases in adverse health effects on the same day or on subsequent days. The episodes generating these acute health effects typically involved very high concentrations both of particles and of SO\textsubscript{2}. The episodic nature of these events, and the concurrent elevation both of particles and SO\textsubscript{2}, underlay the setting of standards for these pollutants jointly. While it was not explicitly stated that no adverse health effects occur because of ambient pollution at concentrations lower than these values, the standards were nevertheless often interpreted as ‘safe’ levels.

Numerous well-conducted epidemiological studies in the past 15 years, and especially since about 1990, have however shown that this sense of safety is not well-founded. There is now a broad-based body of evidence showing small but definite increases in risks associated with increases in air pollution (notably particles and ozone, and plausibly SO\textsubscript{2} also), within the ranges of daily variation typical of the locations studied. These studies come principally from North America but increasingly also from Europe and other countries. They have shown relationships quantitatively linking with health endpoints ranging from premature mortality
through hospital admissions, emergency room visits (ERVs) and restricted activity days (RADs) to exacerbation of asthma, respiratory symptoms and loss of lung function.

Thus, the effects are not restricted to high-episode days but occur at ambient concentrations previously considered to be safe. This evidence strongly suggests that there may be adverse health effects even with the small increments in air pollution associated with modern electricity production. Against this background, it was considered that the public health effects from air pollution caused by power generation constitute a priority impact pathway for the fossil fuel cycles and some other cycles (e.g. waste reclamation).

4.1.4 Acute and chronic effects

It is necessary to distinguish acute effects which occur on the same day as increases in pollution, or very soon thereafter, from the chronic or delayed effects of possible long-term exposure. As described below, acute effects of several pollutants across a range of health endpoints are well established. There is informed speculation but however no established understanding of the mechanisms by which these effects occur.

It is more difficult to establish reliable conclusions in chronic effects studies, and so there are fewer usable exposure-response functions. It may well be however that in terms of public health impacts and their economic valuation, chronic effects are more important. We have therefore attempted to include chronic mortality and morbidity relationships within the overall implementation.

4.1.5 General strategy

The incremental air pollution attributable to power generation is a mixture of pollutants emitted from a power station, and those formed subsequently as the emissions interact with the external environment. Both background levels and the pollution increments vary by time and place; the incremental pollution varying also by technology and fuel source.

Given this complexity, it is not surprising that there are no appropriate studies of the incremental effects on health of those specific pollution mixtures with which this project is concerned. For example, epidemiological studies of adverse health effects in communities around power stations would not provide the kinds of exposure-response (E-R) relationships needed for the present study.

Following the work of Ostro, Krupnick and colleagues in the companion US study (ORNL, 1994), our strategy has been to disaggregate the air pollution mixture and to provide, where possible and warranted, E-R relationships separately for the four principal pollutants: particles, NOx, SO2, and ozone. In its early drafts, the ExternE study drew very heavily on the US work (ORNL, 1994) in terms of its general approach, specific implementation and general feasibility. The present phase of ExternE work, though still strongly influenced by the US study, are based on a separate and more recent evaluation of the available literature. (Working independently, Ostro and colleagues have also updated their approach to quantification in further studies of the costs of ambient air pollution (Ostro, 1994; Rowe et al, 1994)).
The assessment, while conducted for the pollutants and for endpoints individually, has however been an integrated exercise, with decisions and conclusions in any one area impacting on another. This co-ordination has included both strategic judgements; and details of implementation, for example using relationships for one pollutant adjusted for the effects of another. Thus we have tried to assess the evidence in context.

4.1.6 Criteria for E-R functions

For each of these pollutants we have identified, where appropriate, exposure-response relationships describing changes in health endpoints associated with unit changes in pollutant. All the exposure-response relationships proposed are based on epidemiological studies of general air pollution; experimental studies (whether human or animal) are used to provide contextual information only. Within this framework, we have principally restricted consideration to strong studies which show a clear (statistically significant) relationship between pollutant and endpoint of interest, in a well-designed study using appropriate statistical methods, and adjusting suitably for the effects of possible confounding factors, including other pollutants. Specifically, we have attempted to focus on exposure-response relationships that are:

i. Credible as a set of functions (including the additivity or not of estimated impacts across different health endpoints and individual pollutants) against the background of what is known generally about the effects of air pollution;

ii. Reliable individually, i.e. from well-conducted studies, of appropriate design, using appropriate statistical methods and adjusting for confounding factors such as weather and seasonal or other longer-term trends;

iii. Transferable/ generalisable; i.e. from studies in situations that are similar enough to the proposed applications;

iv. Usable within the project: e.g.

- With a health endpoint that can be valued monetarily;
- With exposure characteristic that is compatible with dispersion modelling of incremental pollution; and
- With an E-R relationship that can easily be implemented: ideally, linearised, independent of background levels.

These generally high standards have been modified in some instances where, all things considered, it is likely that a real relationship exists between the pollutant and endpoint of interest; but no particularly strong study of that relationship is available currently. (In these circumstances, use of a somewhat weaker study may be better than no attempt at quantifying that relationship.)
4.1.7 Background review

Epidemiological study of the health effects of ambient air pollution has been a very active area of research in recent years, and is likely to continue to be so in the coming period. Moreover even though, for acute effects in particular, there is now a very substantial body of epidemiological evidence showing quantitative relationships, there is still no well-established understanding of mechanisms, especially for the effects associated with particles. Thus, identifying and implementing a suitable set of exposure-response functions within the present project has meant taking definite positions on issues which are still under active research and are matters of legitimate debate.

The present Chapter considers the evidence and how it has been used in the present project. First we consider the evidence as a whole; and on that basis form a number of strategic judgements or model assumptions which constitute the broad framework of the risk model used for quantification. Because of the scale of available evidence, the basis for these decisions is described reasonably cursorily only. Note that the review focuses on the evidence as available when the strategic judgements were made. For ozone, this was in early summer 1994; and in autumn 1994 for particles, SO$_2$ and NO$_x$. The present Chapter includes several comments and updates in the light of evidence subsequently; but does not presume to review the most recent evidence thoroughly.

Within the framework of those model assumptions, specific exposure-response (E-R) functions are selected which, as a set, are used in quantifying the health effects associated with the incremental pollution from power generation. These E-R functions, and the studies within which they were estimated, are described in some detail. There is, finally, a discussion of the strengths, limitations and uncertainties of the entire quantification effort.

4.2 Acute Effects of PM, SO$_2$, NO$_x$: Model Assumptions

4.2.1 Summary description of the evidence

There is now very substantial evidence from well-conducted epidemiological studies linking daily ambient concentrations of particles with increases in acute health effects over a wide range of health endpoints. There are also numerous reviews of that evidence. An early recent review of acute effects was by Bates (1992) who considered the question of coherence, i.e. of consistency of effect across various health endpoints. Ostro (1993) focused on mortality but considered other endpoints also. Schwartz (1994a) includes a meta-analysis of acute mortality studies. Dockery and Pope (1994) gives a quantitative summary of effects across a wide range of endpoints. Lipfert (1994) and Pope et al (1995a; 1995b) consider chronic effects (i.e. the development of disease related to long-term exposure to pollution) as well as acute effects. The US EPA has recently published a comprehensive report on particles. A review by the UK Department of Health (DH) is in an advanced draft, to be published later in 1995 (COMEAP, in press).
Because of these reviews, most of which are now easily accessible, we do not attempt to present here a further comprehensive assessment of the underlying evidence which continues to grow rapidly. For example, the following summary description includes as yet few results from the EC-sponsored APHEA study of acute mortality and/or hospital admissions in 12 European urban areas (Athens, Barcelona, Bratislava, Cracow, Koln, Lodz, London, Lyon, Milan, Paris, Ponzan and Wroclaw) and co-ordinated by Dr. Klea Katsouyanni in Athens. These results, most of which are pre-publication, will be important in establishing a much better European base for some of the key E-R relationships. Similarly the EC-sponsored PEACE study, co-ordinated by Prof. Bert Brunekreef at Wageningen, the Netherlands, will when completed provide new information on less serious respiratory effects in children, based on panel studies in several European countries.

Following a summary review of the scope and scale of the evidence, we deal more comprehensively with a range of methodological and policy-related questions arising from these studies. These are issues which have a bearing on interpretation of causality and quantification of effects. (The evidence on these questions is assessed in greater detail in the forthcoming UK DH report.)

a) Acute mortality: Well-conducted studies quantitatively linking daily mortality with day-to-day variations in PM air pollution include:

i. Various re-analyses of the mortality and air pollution in London winters, 1958/59 to 1971/72 (Mazumdar et al, 1982; Ostro, 1984; Schwartz and Marcus, 1990; Ito et al, 1993);

ii. A wide range of studies from the USA (Fairley, 1990; Schwartz, 1991a; Schwartz and Dockery, 1992a, 1992b; Dockery et al, 1992; Pope et al, 1992; Schwartz, 1993a; Schwartz, 1994b; Kinney et al, 1995);

iii. Studies from various other European cities: Athens (Touloumi et al, 1994); Erfurt (Spix et al, 1993); Milan (Rossi et al, 1994);

iv. Studies from other cities internationally: Beijing (Xu et al, 1994); Sao Paolo (Saldiva et al, 1995); Santiago (Ostro et al, in press).

It appears that the excess deaths are principally from cardio-respiratory causes in older, ill people and that the associated loss of life may be small on average (see 4.2.7, below). Additional evidence from Europe is provided by recent studies of winter smog pollution episodes which show clear increases in mortality following increases in pollution mixtures including PM (Wichmann et al, 1989; Anderson et al, in press).

b) Hospital Admissions: Lipfert (1993, 1994) and Quenel et al (1994) together give a comprehensive and broad-based review of hospital usage studies, including both hospital admissions and ERVs. The key groups of studies linking hospital admissions for various respiratory causes and ambient PM pollution include:

i. Studies in Southern Ontario of the relationships between hospital admissions for asthma and ‘summertime haze’, including sulphates, acidity and ozone (Bates and Sizto, 1983;
1987; 1989; Lipfert and Hammerstrom, 1992). Recent studies in all Ontario (Burnett et al., 1994) and in the Toronto area specifically (Thurston et al., 1994a, 1994b) have examined a wider range of respiratory conditions, the latter seeking to differentiate the effects of various measures of ambient PM pollution.

ii. Studies in New York State (Thurston et al., 1992), where pollution mixtures are similar to those in Southern Ontario.

iii. Studies in Utah Valley (Pope, 1989; 1991) where the principal source of particulate pollution was a steel mill. The hypothesis that these findings are artefactual, with increased childhood hospital admissions due rather to viral infection (Lamm et al., 1994), does not appear to be well-founded (Lipfert, 1994).


v. In Europe, recent published studies have included positive findings from Birmingham (Walters et al., 1994) and findings from Helsinki reporting associations with SO$_2$ and NO$_x$ but not with particles (Ponka and Virtanen, 1994). The APHEA results will extend this base very substantially.

c) Emergency Room Visits (ERVs): The key studies linking ambient PM pollution with ERVs for various respiratory conditions include:

i. An early study in Steubenville, Ohio (Samet et al., 1981) which does not clearly separate out an effect of PM from a possible effect of SO$_2$;

ii. Emergency room visits for chronic obstructive pulmonary disease (COPD) in Barcelona (Sunyer et al., 1991; 1993).

iii. ERVs for asthma in Vancouver (Bates et al., 1990) and in Seattle (Schwartz et al., 1993). Another study, in North-East USA, found relationships between ERVs for asthma and ozone, but not PM (Cody et al., 1992).


d) Restricted Activity Days (RADs): Two papers using data from the US Health Interview Survey (HIS) report relationships linking various forms of restricted activity day with particulate air pollution (Ostro, 1987; Ostro and Rothschild, 1989). A third paper describes relationships with ozone but not with PM (Portney and Mullahy, 1986).

e) Provocation or exacerbation of asthma: Evidence of the provocation or exacerbation by air pollution of asthma attacks in persons already identified as asthmatic comes from panel studies which variously examine increases in medication (bronchodilator) usage and/or a complex of respiratory symptoms and lung function changes which constitute an ‘asthma attack’. Two of the key studies quantifying effects are Whittemore and Korn (1980) and Ostro et al., 1991.
f) *(Respiratory) symptoms:* In these studies, lower respiratory symptoms are usually taken as wheeze, dyspnoea or chest tightness/discomfort. Cough has been included in some studies while being separated out in others. Upper respiratory symptoms are limited to nasal symptoms and sore throat. The key studies examining symptoms include Krupnick *et al* (1990); Pope and Dockery (1992); Ostro *et al* (1993); and, in Europe, Braun-Fahrlander *et al* (1992); Roemer *et al* (1993); Hoek and Brunekreef (1993; 1994). Schwartz *et al* (1994) is a major study of symptoms in children unfortunately too late to be included among the E-R functions used for the present assessment.

g) *Lung function:* Lung function studies have focused on peak flow, though several have examined changes in spirometric variables (FEV<sub>1</sub>, FVC) also. As for symptoms, there has been a particular focus on panel studies of air pollution effects in children, though several studies examine adults also. Lung function effects are not amenable to economic valuation currently and so are included only peripherally in the present evaluation.

### 4.2.2 Are the key studies reliable?

(a) *Reliability of design:* The various study designs used in air pollution epidemiology have been reviewed at a series of Workshops in 1992/93 of the EC COST 613/2 Concerted Action on Air Pollution Epidemiology (CEC, 1993). That report also contains a useful assessment of study validity and potential biases and a brief review of methodological considerations for meta-analyses.

Almost all of the epidemiological studies of acute effects considered here are of time series/longitudinal design. This design has the advantage that, by studying much the same population over time, it safeguards against confounding with population characteristics (lifestyle, occupational exposures, smoking, diet, levels of chronic disease etc.). The most noteworthy exceptions were the HIS studies of RADs which were cross-sectional in design and so are less protected against confounding.

(b) *Reliability of health effects:* The reliability of health effect assessment in air pollution epidemiology has been considered recently both by a Working Group of the EC COST 613/2 Concerted Action on Air Pollution Epidemiology (CEC, 1991a) and by Samet and Speizer (1993). In general, reliable characterisation of health effects is relatively straightforward. For example, records of daily deaths by defined geographical region are good in the Western industrialised countries. Cause of death, though subject to error, is well established as an epidemiological endpoint.

There were doubts initially about the adequacy of reporting of hospital usage data for use in epidemiology (Bennett, 1981). However, Lipfert (1993) highlights that hospital data ‘tend to be less subjective than symptom data; and.. unlike mortality, a trip to the hospital is not an inevitable outcome and thus the degree of prematurity is not an issue.’ Detailed diagnoses, for example classification as ‘asthma’ rather than as ‘bronchitis,’ is to some extent arbitrary (Bates and Sizto, 1987). There is however some evidence (Delfino *et al*, 1994) and understanding (Quenel *et al*, 1994) that such classification errors are likely to be unimportant in practice.
The reporting of restricted activity days (RADs) in the US Health Interview Survey (HIS) is more suspect: assessment of restricted activity is inherently subjective, and that assessment was sometimes done by an informant other than the study subject (Portney and Mullahy, 1986). However, the HIS data are but a small part of the overall picture.

Although there is little clear information on the reliability of the data acquired from panel or event studies, there are differences in reliability according to the outcome being measured. Whittemore and Korn (1980) suggested that objective measures of responses should be made rather than less clearly defined subjective assessments such as ‘asthma attacks’. Symptom reports or scores are also inherently subjective, though these constitute an important health endpoint of panel studies.

As well as reliability of endpoint in studies individually, there is also the question of consistency across studies. For example, there may be differences between studies, between regions, and between subgroups within studies, in how that endpoint is assessed. Such differences complicate comparisons of relative risks across studies; and transferability of findings.

The importance will vary according to endpoint studied. The measurement of more-or-less objective characteristics such as cause-specific mortality or lung function should be reasonably independent of location and so highly transferable. Respiratory symptoms may be more location-specific, given that symptoms are based on self-assessment. Hospital usage data and RADs are likely to be the most location-specific of the various health endpoints considered, because these depend on the organisation of and access to major institutions such as health service, work and education. Against this background, it seems likely that quantitative estimates of effects are more reliably transferable between locations if expressed as percentage change (per unit of PM exposure) rather than as absolute numbers.

Finally, it is most likely that any random errors in assessing health effects will tend to weaken the power of the studies to detect any real relationships with air pollutants, by reducing statistical significance. It is very highly unlikely that they will tend to generate positive relationships artefactually.

(c) Reliability of exposure characterisation: For a general review of methodological aspects of the reliability of exposure estimation in air pollution epidemiology see for example the EC report (CEC, 1991b). Almost all of the studies considered here use daily measurements of ambient concentrations from fixed-point monitors, rather than measurements of personal exposures, to represent the effect of ambient air pollution. In the context of these studies it makes sense to consider four aspects of reliability.

i. Reliability of fixed-point monitoring: Measurement methods for particles (PM$_{10}$) are now well-established and reliable, as are automated techniques for measurement of SO$_2$ and other gases. In particular, it does not seem likely that unreliability in measurement of other pollutants, for example SO$_2$ or NO$_2$, is a reason why relationships have been identified with particles rather than the gases. The agreement between different monitors located in much the same broad study region is discussed in several studies and found to be good. The
impact on estimated risks of choice of monitor or of averages over monitors is examined by Ito et al. (1995).

ii. Use of PM$_{10}$ or TSP as a common metric across studies: Quantification of the effect of particles requires use of a common metric such as PM$_{10}$ (Ostro, 1993; Dockery and Pope, 1994), or TSP (Schwartz, 1994a). This requires the use of conversion factors which are necessarily approximate. For example, the convention that PM$_{10}$ constitutes about 0.55 TSP seems to have good support, at least for studies in the USA. On the other hand, the suggestion that British Smoke is comparable to PM$_{10}$ is much less realistic, the relationship varying markedly by pollution sources.

iii. The relationship of fixed-point to personal exposures: Most of the studies use ambient fixed-point measurements. The personal exposures of individuals studied are however more relevant biologically. More good studies are needed to give reliable information about the relationships between (average) personal and (average) ambient exposures to different pollutants in the regions studied. This might include further study of ambient and indoor concentrations of pollutants; and information about individuals’ time-activity patterns in various micro-environments. Variations in the relationships between personal and ambient (fixed-point) exposures would adversely affect the ability to detect relationships and to extrapolate results reliably.

iv. The variability of personal exposures: Clearly, the personal exposures of individuals vary about their average value. One implication is that population studies using average ambient exposures can appear to show no threshold for effects even when there is such a threshold for every individual (J. Cherrie, personal communication). Another is that risk estimates are affected by imprecision in exposure estimates, and indeed in confounding factors also, even when bias has been avoided (Armstrong, 1991; Thomas et al., 1993; Lipfert 1994). The effect of measurement error in a single exposure variable, when other exposures and confounders are measured precisely, is to lead to underestimation both of risks and of their statistical significance. The direction of bias can vary with errors in several variables simultaneously.

(d) Reliability of statistical methods: The three main aspects are distributional issues (i.e. representing random variation); forms of representing confounding factors; and adjusting for (residual) autocorrelation.

i. Distributional assumptions: Statistical regression modelling seeks to explain the average or expected value of an outcome variable by explicitly representing its dependence on one or more explanatory variables. For example, expected number of daily hospital admissions in the population of a specified region on a given day might be represented in terms of dependence on season, week-day, weather and air pollution characteristics. This is called the ‘systematic’ or ‘explained’ part of the regression model. Estimation of this systematic component depends in part on the real and assumed pattern of ‘unexplained’ or random variation also present in the data. So, in principle, the distributional assumptions made in air pollution epidemiology may influence risk estimates.
Regression methodology was developed in the context of ‘normal’ or Gaussian random error which is often appropriate where the outcome variable is continuous (e.g. lung function); and for counts, where daily numbers are large. Regression methods have in the past 30 years or so been extended to include such aspects as logistic regression of binary outcomes (e.g. individuals’ reports of presence/absence of a symptom on a given day) and Poisson regression analyses of counts (e.g. numbers of daily deaths or hospital admissions). These approaches are particular instances of generalised linear modelling (McCullagh and Nelder, 1989) which is now well-established in statistical theory and practice. Many of the studies of daily numbers of events use Poisson regression analyses (possibly allowing for overdispersion).

On general theory grounds Poisson regression is the most appropriate approach when analysing counts, though many studies use ‘normal’ distributional assumptions also. Fortunately, it appears that the estimated effects are not sensitive to exactly how random variation is represented. For example, Kinney et al (1995) examined the role of PM pollution within three different kinds of regression model: ordinary least squares; log-linear regression (i.e. exponential form for expected mortality, with normally distributed errors proportional to level of mortality); and Poisson regression. They reported no difference in relative risks linking PM$_{10}$ with acute mortality as estimated from the three models.

ii. Statistical methods when adjusting for confounders: Seasonal and other longer-term time patterns can be accommodated within Poisson regression for example by use of indicator (dummy) variables, as in Schwartz (1994d). Alternatively, these longer time patterns can be removed explicitly, for example by subtracting a running mean, or similar moving average, prior to further modelling. This usually leads to a non-integer response and analysis using Gaussian random error. One such filter, developed during analyses of London data (Shumway et al, 1983) uses a weighted mean over 19-day periods centred on the day in question, with the weights being progressively smaller for days nearer the endpoints of the 19-day period and has been used by many other researchers (Kinney and Ozkaynak, 1991; Bates and Sizto, 1987; Thurston et al, 1992, 1994b; Schwartz, 1995). This might be considered an example of generalised additive modelling (Hastie and Tibshirani, 1990), an approach which avoids the need to specify the shape of the relationship between confounding factor and health outcome.

iii. Autocorrelation: Similarity of outcome on adjacent or proximate days is an issue in studying health effects over time. Such autocorrelation may to some extent be explained by similarities in explanatory variables such as season, weather and pollution. Unexplained or residual autocorrelation may however remain even after adjustment for these factors; and, according to its severity, may distort risk estimates (specifically, may inflate estimates of statistical significance) if unaccounted for in the analysis.

Methods of handling residual autocorrelation in ‘ordinary’ linear regression modelling have been well-established for many years. The Poisson regression air pollution analyses of Schwartz and colleagues use a more recent methodology of generalised estimating equations (GEE) which extends conventional Poisson regression and other generalised linear modelling to take account of residual/ unexplained autocorrelation between days
(Liang and Zeger, 1986; Diggle, Liang and Zeger, 1994). Moolgavkar et al (1995) consider that the GEE approach of Liang and Zeger may be inappropriate in the population-based time series analyses because of small sample size. The issue may best be considered as open at present. Fortunately, however, it seems not to be of great practical importance because residual autocorrelation in most studies is low and results are usually insensitive to adjustment for autocorrelation or not. In practice, therefore, conclusions are based on generalised linear models without use of the more recent GEE methodology.

iv. Conclusions: The statistical methods used in the principal papers are reliable and appropriate; and there is no substantial basis for considering that the positive associations reported are an artefact of the relatively sophisticated statistical methodology often used.

4.2.3 Is adjustment for (non-pollution) confounders adequate?

The broad strategy which Schwartz and co-workers adopt to the modelling of confounders within a Poisson regression is first, develop a baseline model which seeks to explain as much as possible of the daily numbers of deaths (or admissions) in terms of those non-pollution confounders which can be studied; then, establish whether this baseline model can be improved by also including measures of particles and other air pollutants; and finally, examine whether the estimated effects of particles are sensitive to the exact representation of the non-pollution confounders; i.e. whether changing the baseline model leads to changes in the estimated effects of pollution.

This is a sound strategy in dealing with confounding factors, especially when the baseline model overfits, rather than under-represents, the effect of non-pollution variables. Also, sensitivity analyses within studies generally show that the estimated effects of air pollution are insensitive to the specific representation of non-pollution confounders. This supports the view that the relationships with air pollution are not artefacts of unadjusted confounding factors.

There remains, however, the issue of unmeasured confounders and how they are dealt with. Some of these, for example, flu episodes, may show as residual autocorrelation between days which is dealt with in the analysis (see above). Possibly more important is the question of whether effects of weather have been taken sufficiently into account. These are, typically, much more important than pollution in terms of explaining day-to-day variations in mortality (Kalkstein 1993). Also, temperature effects on daily mortality and hospital admissions are markedly non-linear, in the sense that adverse effects are found under extremes of heat and cold rather than under ‘normal’ conditions (Alderson 1985; Kunst et al, 1993; Touloumi et al, 1994). In general, air pollution studies have taken due account of this non-linearity when adjusting for temperature effects.

Another aspect is that the effects of temperature on mortality may be both delayed and complex. Thus Kunst et al (1993), studying daily mortality in The Netherlands 1979-87, and adjusting for influenza incidence and season, examined direct and indirect weather effects for periods of up to one month. The identified effects of moderate ‘heat’ (increases above 16.5°C) were rapid, and within one week. The principal effects of moderate ‘cold’ (decreases below 16.5°C) were also rapid, with some delayed effects mediated via influenza. The confounding
of weather with air pollution was examined in some detail subsequently (Mackenbach et al., 1993). Same-day association between SO₂ and mortality in The Netherlands 1979-87 disappeared when examined in a baseline model that included complex lagged temperature effects, especially ‘cold’ lagged 1-5 days. The pollution data in this study were, however, quite crude: no measures of PM were examined, and a single set of daily SO₂ values was applied to The Netherlands as a whole.

A further complexity is that adjustment using the standard measures of weather (e.g. temperature, relative humidity) may not be sufficient. For example, the Dutch studies reported interactions between temperature, wind speed and humidity in describing the effects of weather on mortality (Kunst et al, 1993). Or, it has been suggested that oppressive night-time conditions following a very hot day may be more stressful than the maximum temperature as such (Kalkstein and Davis 1989; Kalkstein 1991). Consequently, it has been proposed that the effects of weather on mortality be described using a ‘synoptic’ approach (Kalkstein 1991, 1993). This allows for the simultaneous evaluation of numerous weather elements by combining them into groups or categories that are representative of the variety of micro-climates in a given location; and examining mortality difference in terms of these synoptic categories or micro-climates. This approach may help to distinguish better between weather and pollution effects in future studies of air pollution and acute mortality.

Thus, even in studies where weather effects have been modelled with care, residual effects of weather may remain, unadjusted. If these are also related to pollution, they might be a source of artefactual relationships between pollution and mortality or morbidity (Waller and Swan 1992, Mackenbach et al 1993, Kalkstein 1993).

It is important therefore to review the evidence across studies. Here, the most striking aspect is the variety of situations studied, and, in particular, the range of different dependencies between temperature and pollution covered by the major mortality studies. Thus, for example, the climatic conditions and their interactions with pollution encompassed by studies in London, Athens, Detroit, Birmingham (USA), Utah, Missouri, California and Santiago is diverse in terms of seasonal dependencies and humidity. Nevertheless, the estimated effects of PM on acute mortality in these diverse situations are strikingly similar. The argument advanced by Schwartz and co-workers seems convincing: that whereas a case might be made for inadequate control for weather in a specific location, it is practically inconceivable that such mis-specification would consistently result in an apparently positive effect of PM pollution, of broadly similar magnitude, across a range of different temperature/pollution conditions.

In conclusion, then, the principal population-based studies have adopted a fundamentally conservative strategy in attributing mortality or hospital usage effects to air pollution in the presence of weather and other confounders and this strategy has been implemented carefully. The effects of weather on mortality in particular are, however, sufficiently important, sufficiently complex, and sufficiently confounded with pollution that there is room for legitimate doubt that pollution effects might be distorted. However, mortality studies across a wide range of locations with differing climates and differing patterns of correlation between climate and pollution show similar relationships between daily ambient PM and acute
mortality. This similarity of effects across different conditions gives reassurance that the apparent effect of air pollution on mortality is not an artefact of incomplete adjustment for weather. Relationships of air pollution with other endpoints such as symptoms and lung function are less likely to have been affected by climate.

4.2.4 Which pollutant? In particular, do we need to consider pollutants other than particles?

(a) General remarks on mixtures and interactions: As noted earlier, ambient air pollution is experienced as a mixture of aerosols and gases; and it is not straightforward to attribute its apparent diverse health effects to the individual constituents of the mixture. The weight of epidemiological evidence however points strongly towards particles as the pollutant with which quantitative relationships have been found most consistently. Our aim in the present project is to develop a risk model which is reasonably simple, sound and applicable. We decided therefore that relationships of health effects with particles would form a core of the quantification to be implemented.

This strategic decision led to a number of important questions regarding other pollutants; for example: Can the role of particles be distinguished reliably from that of other pollutants? Do other pollutants have a quantifiable effect over and above that of particles? If so, is it additive to that of particles or is there some synergism which should be represented? The present section addresses these questions in respect of \( \text{NO}_x \), \( \text{SO}_2 \) and ozone. Later sections consider detailed aspects of the relationships between particles and acute health effects.

(b) Particles and \( \text{SO}_2 \): Existing WHO Air Quality Guidelines for particles have been based principally on winter-time pollution episodes (‘winter smog’) where elevated levels of PM are accompanied by high levels of \( \text{SO}_2 \) also. Consequently, standards have been set jointly for particles and \( \text{SO}_2 \). Studies of another recent European pollution episode, in Germany in 1985 (Wichmann et al, 1989), support these broad findings but do not help in separating the effects of \( \text{SO}_2 \) and PM.

There however been attempts at distinguishing the respective contributions of PM and \( \text{SO}_2 \) using data from London winters 1958/9 to 1971/2. These have provided evidence implicating PM (measured as Black Smoke) rather than \( \text{SO}_2 \) (Mazumdar et al 1982; Schwartz and Marcus 1990). Following questions on the grounds of biological plausibility, further analyses supported the associations with smoke rather than \( \text{SO}_2 \) (Schwartz, 1991) and did not suggest that the PM effect was modified noticeably by \( \text{SO}_2 \). However, preliminary findings from yet more analyses of London data, 1965-72, using a novel method of adjusting for season and weather, have pointed to acidity (\( \text{H}^+ \)) and \( \text{SO}_2 \) rather than to PM (black smoke) as being principally implicated (Lippmann and Ito, 1995).

Some of the recent North American studies reporting relationships between PM and acute mortality were conducted in areas where \( \text{SO}_2 \) levels were uniformly low (Fairley 1990; Pope et al, 1992) and consequently \( \text{SO}_2 \) was not examined directly. These studies show that presence of \( \text{SO}_2 \) is not a pre-condition of a PM effect. Some other studies, where particles and \( \text{SO}_2 \) were both examined, report relationships with PM but not with \( \text{SO}_2 \). These include studies of mortality (Dockery et al, 1992) and of hospital admissions (Thurston et al, 1994b).
In both these studies, relationships with health were found for sulphates but not for SO$_2$ as a gas.

Other US studies report relationships linking acute mortality with both PM (TSP) and SO$_2$ when each is included separately in Poisson regression models, adjusting for other factors; but find that, when both TSP and SO$_2$ are jointly included in these analyses, the relationship of mortality with TSP is little changed, whereas that with SO$_2$ is seriously reduced (Schwartz and Dockery, 1992a; 1992b). The authors, probably correctly, interpret this pattern as evidence that the role of PM is more fundamental than that of SO$_2$ in these studies. Recent re-analyses of the Steubenville mortality data (Moolgavkar et al., 1995) imply that the findings pointing to TSP rather than SO$_2$ may not be well-founded in these data. These re-analyses still find a relationship between mortality and TSP, but a weaker one than before. Further analyses of these and other US studies are being sponsored by the US Health Effects Institute (HEI) and are due to be reported at the ISEE/ISEA Conference in the Netherlands, August 30 - September 1, 1995.

Many of the key European studies have used Black Smoke as the principal measure of particles. Results often show relationships with both black smoke and SO$_2$, when both are included in analyses adjusting for longer-term fluctuations and for weather (Touloumi et al., 1994; Sunyer et al., 1993). Note that earlier analyses from these locations had pointed to SO$_2$ rather than to smoke (Hatzakis et al., 1986; Sunyer et al., 1991). The French two-cities study reported relationships of mortality with SO$_2$ but not with smoke (Derriennic et al., 1989). On the other hand, the Birmingham hospital admissions study found relationships with both smoke and SO$_2$ examined separately, but principally with smoke when both pollutants were examined together (Walters et al., 1994).

Two recent studies appear to contradict the general evidence favouring PM rather than SO$_2$ as an influence on mortality and hospital admissions. A study of daily mortality in Beijing showed relationships with SO$_2$ in both summer and winter; but with TSP in summer only (Xu et al., 1994). Pollution, principally from domestic fires and industry, was generally very high with the median TSP being 336 µg/m$^3$ for TSP and 40 µg/m$^3$ for SO$_2$. TSP concentrations were however strongly influenced by natural soil dust, especially in winter and spring; and so daily SO$_2$ may more accurately reflect pollution from combustion sources, including PM, than does daily TSP.

Similarly, a study of hospital admissions in Helsinki reported relationships with SO$_2$ and NO$_x$, but not with TSP or ozone (Ponka and Virtanen, 1994). Again, the TSP measurements were strongly influenced by dusts from natural sources. The authors ascribe the relatively high mean concentration of 76 µg/m$^3$ TSP to the ‘meteorological conditions and the erosions of street surfaces caused by studded tyres during the winter, as well as the use of sand on the streets to treat icy surfaces’. The principal sources of SO$_2$ and of NO$_x$ were industry and traffic, respectively. It is again reasonable to look on SO$_2$ and NO$_x$, rather than TSP, as markers of fine particulate pollution from these combustion sources. Thus, it may well be that the observed relationships in these two studies reflect particulate pollution indicated by SO$_2$ and/or NO$_x$, rather than an effect of gases as such.
A major new series of analyses on the acute effects of air pollution on mortality and/or hospital admissions from 12 European cities has been carried out in recent years within the EC-sponsored APHEA project. Results, also to be presented at the ISEE/ISEA Conference, suggest a relatively weak but consistent and statistically significant effect of air pollution on daily deaths (Katsouyanni et al., 1995), with relative risks for 50 µg/m³ SO₂ greater and more significant than those associated with suspended particles (Zmirou et al., 1995).

In summary, studies available during late 1994, when strategic decisions for the present project were made, provide strong evidence that a principal role for particles can be identified across a wide range of situations where SO₂ is present in varying concentrations. Overall, the evidence supported a view that the role of PM is more fundamental than that of SO₂; and that the joint presence of SO₂ is not a pre-condition of a PM effect. It was unclear whether or not there is an independent but subsidiary effect associated with SO₂; and, if so, whether SO₂ was a surrogate for other pollutants. For example, the evidence favouring SO₂ was strongest in studies where PM was measured as black smoke. It is possible that in these studies, SO₂ may be acting as a marker for aspects of particles not measured by smoke.

Clearly, in view of the new APHEA results, the situation must be kept under review.

(c) Particles and NOₓ: Relatively few epidemiological studies report exposure-response relationships linking ambient NOₓ with mortality or morbidity. In those that do, PM is generally also implicated; or NOₓ is arguably a surrogate for unmeasured or inappropriately measured PM. Thus, findings of increased daily mortality associated with daily NOₓ (Kinney and Ozkaynak, 1991) could very reasonably be ascribed rather to particulate matter represented as KM (a measure of particulate loading based on optical reflectance and similar to Black Smoke), with which NOₓ was highly correlated in these data. Similarly, increased mortality during the December 1991 London pollution episode, which initially appeared as an episode of NOₓ, could equally be attributed to PM which was also elevated on those days (Anderson et al., in press). The only mortality data apparently implicating NOₓ, without strong evidence for PM as an alternative explanation, are recent preliminary results of childhood mortality in Sao Paulo (Saldiva et al., 1994), where further work is in progress to investigate the apparent absence of a PM effect. (PM, rather than NOₓ, is related to daily deaths among the elderly in Sao Paulo: Saldiva et al., 1995.)

Similarly for morbidity endpoints. Thus, as noted above, relationships of hospital admissions with NOₓ in Helsinki (Ponka and Virtanen, 1994) may reasonably be understood as an effect of traffic pollution generally, including combustion PM from traffic. Quantitative associations between NOₓ and morbidity in a Norwegian panel study are clearly interpreted by the authors as representing an effect of general traffic pollution (Clench-Aas et al., 1991) rather than an effect of NOₓ as such. (Other pollutants, including PM, were not measured directly in that study.) Relatively recent re-analyses of the Los Angeles student nurses panel data also found relationships between NOₓ and eye irritation and some respiratory symptoms (Schwartz and Zeger, 1991). Though other pollutants were examined in this study, these did not include PM; and so these results are not dependable evidence of an independent effect of NOₓ on morbidity.
There remain occasional instances of what appears to be a small NO\textsubscript{x} effect additional to that of PM (Braun-Fahrlander \textit{et al}, 1992; Schwartz \textit{et al}, 1991). Overwhelmingly, however, the epidemiological studies show an effect of PM independently of NO\textsubscript{x} levels, whereas most of the relatively few positive findings associating NO\textsubscript{x} with acute health effects may simply reflect a PM effect. On that basis, we made no attempt to quantify an effect of NO\textsubscript{x} additional to that of particles in the present project.

\textit{(d) Particles and ozone:} In studies which examine mixtures of pollutants, the confounding of PM with ozone is generally less serious than the confounding of PM with SO\textsubscript{2} or NO\textsubscript{x}. Several studies report exposure-response relationships linking daily variations in O\textsubscript{3} with variations in daily mortality (Kinney and Ozkaynak, 1991); hospital admissions (Thurston \textit{et al}, 1992, 1994b; Schwartz 1994c, 1994d, 1994e, 1995); ERVs (Cody \textit{et al}, 1992); restricted activity (RADS) (Portney and Mullahy, 1986; Ostro and Rothschild 1989); asthma attacks (Holguin \textit{et al}, 1985); respiratory symptoms (Krupnick \textit{et al}, 1990; Ostro \textit{et al}, 1993; Schwartz \textit{et al}, 1994) and lung function. Indeed, many of the summer panel studies reviewed by the UK Department of Health (MAAPE, 1991) showed no effect of particles but, more often than not, an effect of ozone on lung function. Most of the other studies found relationships with PM also, even in regression models which simultaneously included both PM and ozone. It appears that in these studies, neither PM nor O\textsubscript{3} is acting as a surrogate for the other.

On this basis, we considered that a full review of the acute effects of ozone was warranted, with a view to including estimates of ozone effects as well as those of particles. The results of that review are given in the next Section (4.3).

\textbf{4.2.5 What is the appropriate index of particles?}

Particles vary both by size and composition; and it is recognised that their effects upon health may vary accordingly. However, epidemiology is of limited value in attempts at identifying the aspects most related to disease, because the various indices are usually highly correlated within a location or region. Thus, when one index only is examined, it may be acting as a surrogate for other characteristics of PM pollution; and when several indices are examined, it may be practically impossible to attribute effects independently.

\textit{(a) Studies examining principally one index of PM pollution:} Many studies examine at most one measure of particulate air pollution, dictated by what measurements are made routinely. Thus, UK studies and many studies elsewhere in Europe have used black smoke only (e.g. Schwartz and Marcus, 1990; Walters \textit{et al}, 1994; Touloumi \textit{et al}, 1994; Sunyer \textit{et al}, 1993; Wichmann \textit{et al}, 1989). Some US studies also use measures of visibility as a surrogate for levels of particles (Kinney and Ozkaynak, 1991; Krupnick \textit{et al}, 1990; Ostro \textit{et al}, 1993).

Other recent European studies use TSP (Schwartz \textit{et al}, 1991; Braun-Fahrlander \textit{et al}, 1992; Ponka and Virtanen, 1994) which has also been widely used as the preferred measure in many studies in the USA and in some other countries, e.g., Beijing, China (Xu \textit{et al}, 1994). It is the common exposure metric to which Schwartz (1994a) converts various studies of acute mortality in comparing their relative risks.
Other recent reviews (Ostro 1993; Dockery and Pope 1994) have expressed the percentage increase in acute health effects in terms of PM$_{10}$ concentrations; and PM$_{10}$ is now the preferred measure in many recent studies, notably in the USA. These include both the population-based and panel studies by Pope and co-workers in Utah Valley; other studies of acute mortality in the US (Schwartz, 1993; Dockery et al, 1992) and elsewhere (Saldiva et al, 1995); studies of hospital service usage (Schwartz, 1994c; 1994d; Schwartz et al, 1993) and panel studies by Brunekreef and colleagues in The Netherlands.

There are also many studies which focus on the smaller size fractions and/or on particular components of the PM$_{10}$ mixtures. These choices reflect the authors’ views of biological plausibility as well as pragmatic considerations of data availability. Thus, the series of hospital admissions studies in Southern Ontario, Canada by Bates and Sizto (1987; 1989) and the extension by Burnett et al (1994) to all hospitals throughout Ontario, examine sulphates as the principal index of PM; with some reference to acidity [H$^+$] also. The associated papers by Thurston et al (1994a, 1994b) will be discussed in more detail below. Similarly, the hospital admissions studies carried out by Thurston et al (1992) in North East USA use sulphates as a principal measure of PM, though H$^+$ is examined also. Ostro et al (1991), in studying shortness of breath in asthmatics, also consider both SO$_4$ and H$^+$. Fine particles (FP), i.e., PM$_{2.5}$ was the measure of choice in analyses of the US Health Interview Study (Ostro 1987; Ostro and Rothschild 1989).

Most of these studies show statistically significant exposure-response relationships linking the particulate measure chosen and the acute health effect endpoint studied. This finding should not be interpreted as proof that each index is biologically relevant. Rather, it may in practice reflect the high correlation between various measures.

(b) Studies comparing different indices of PM pollution: Despite the degree of correlation between the various indices, a few studies have examined which of the various indices of particles is most strongly associated with acute health effects, having adjusted for confounders. Thus, Dockery et al (1992) studied daily mortality and air pollution in St Louis, Missouri and Kingston/Harriman, Eastern Tennessee over a 12-month period 1985-86, with the specific aim of examining the relative importance of various measures of particles and other pollutants, adjusting as usual for non-pollution confounders. The strength of association was assessed in terms of statistical significance. Ranking indices of particulate pollution in this way gave, for the St. Louis data, PM$_{10}$ > PM$_{1.5}$ > SO$_4$ > H$^+$. In Kingston/Harriman, where none of the four measures was statistically significant, the corresponding ranking was PM$_{2.5}$ > PM$_{10}$ > SO$_4$ > H$^+$. The short study period and the relatively small populations at risk (especially Kingston/Harriman) limit the power of this study to detect relationships between daily mortality and PM pollution and to distinguish between indices. Aware of these and other limitations, the authors nevertheless note the lack of evidence pointing towards any measured component of PM$_{10}$ rather than to PM$_{10}$ itself.

Thurston et al (1994a; 1994b) studied summertime respiratory hospital admissions in Toronto, Southern Ontario, over six-week periods during July and August, 1986-88. Several indices of particles were studied, including TSP, PM$_{10}$, FP (PM$_{2.5}$), coarse particles (CP: PM$_{10}$-PM$_{2.5}$), SO$_4$ and H$^+$. The correlations between pollutants, de-trended and adjusted for day-of-
the-week effects, are reported in detail. They include $r(H^+, SO_4) = 0.82$; $r(SO_4, FP) = 0.84$; and $r(FP, PM_{10}) = 0.97$, illustrating the difficulties of identifying effects separately (Thurston et al, 1994a). Adjusting for non-pollution confounders, the clearest relationship with summertime respiratory admissions was found for ozone (Thurston et al, 1994b). Adjusting for ozone also, and fitting the various particulate measures one at a time, the data showed a ranking of statistical significance as $H^+ > SO_4 > FP > PM_{10} > TSP$, with only $H^+$ statistically significant at the 5% level; i.e., much the opposite of that found by Dockery et al, 1992.

(c) Summary/Conclusions: Against this background, some authors are in favour of PM$_{10}$ as the index of particulate pollution most relevant to biological mechanism and/or to public health policy. Reasons for choice of PM$_{10}$ include the diversity of situations and sources of particulate pollution where quantitative exposure-response relationships have been identified; and the similarity of relative risks across those situations when expressed in terms of PM$_{10}$. Whatever the truth about causality, this relative consistency of effect, expressed as percentage change per unit PM$_{10}$, is helpful in any attempts to quantify effects.

Others, e.g., Thurston et al (1994b), are less convinced. They base their reasoning on the biological implausibility of similar effects of the various components of PM$_{10}$, expressed in mass terms; as well as on epidemiological and (limited) experimental evidence. They argue, surely correctly, for full characterisation of PM$_{10}$ mixtures so that the issues can be investigated further. In principle, such characterisation should consider measures of particle count and surface area, as well as mass, on grounds of possible mechanism of effects.

In summary, therefore, the epidemiological evidence regarding the optimal measure of particulate matter is at present inconclusive. Relationships with PM$_{10}$ are well-established and appear convincing, statistically. Alternative measures involving fine particles appear to be more plausible biologically (Seaton et al, 1995) and have some support in epidemiology, but that support is by no means compelling. The answer may not be simple. It is possible, for example, that the true particulate ‘driver’ of the relationship varies by location and/or by health effect; so that simple generalisations may not apply.

4.2.6 Is the (apparent) effect of PM$_{10}$ independent of background levels of PM$_{10}$?

(a) Evidence of a threshold? By this we mean a threshold for the population as a whole. As discussed further below, there clearly is a threshold for individuals, in that only those whose health is already severely compromised are at increased risk of dying on high air pollution days.

Current air pollution guidelines, derived from high pollution winter smog episodes, embody a threshold concept at the population level for PM and SO$_2$. Indeed, as recently as 1982, re-analyses of the London winter mortality data 1958/59 to 1971/72 were used to suggest a threshold of 300 µg/m$^3$ black smoke (Mazumdar et al 1982), though the paper reported positive exposure-response relationships from three winters (1968/69, 1970/71 and 1971/72) when smoke levels were always below 300 µg/m$^3$. It is not surprising, therefore, that further regression analyses of these same data showed no evidence of a threshold (Ostro 1984; Schwartz and Marcus, 1990). Specifically, Ostro split the London winter data into two parts: days when smoke was less than 150 µg/m$^3$, and days when it was greater; and analysed data
from each of the 14 winters separately. There were very few low smoke days in the two earliest years. In each of the remaining 12 years, there was a positive relationship between smoke at 150 µg/m$^3$ or less and daily mortality; that relationship being statistically significant in most years.

This finding, of acute health effects associated with ambient particles even when background levels of pollution are low, is now widespread across locations and health endpoints. For example, in the Philadelphia study (Schwartz and Dockery, 1992b), the 5th percentiles of the TSP distribution were 37 and 132 µg/m$^3$, respectively; i.e., PM$_{10}$ was less than 100 µg/m$^3$ on most, possibly on all, days studied. In Dockery et al (1992) the range of PM$_{10}$ pollution was 1-97 µg/m$^3$ in St Louis, and 4-67 µg/m$^3$ in Eastern Tennessee. In the Detroit hospital admissions paper (Schwartz 1994d), the 10th and 90th percentiles of daily PM$_{10}$ were 22 and 82 µg/m$^3$, respectively. There are many more examples. And within studies, as in Schwartz (1994), the authors often report analyses excluding the highest pollution days (using, for example, 150 or 100 µg/m$^3$ TSP or PM$_{10}$ as cut-off) and find unchanged relationships; and/or examine the shape of the relationship, and find no evidence pointing to a threshold.

Thus, the London data and the cumulative evidence on this issue are consistent. The relationships between daily mortality and daily particulate pollution are not due principally to the effect of some especially high days (‘episodes’). Rather, they reflect day-to-day variations in ‘normal’ pollution levels, well below what previously might be considered to be a threshold.

This cumulative evidence within and across studies does not, indeed cannot, prove that there is no threshold. There is at present a lack of direct information about whether severe adverse effects (mortality, hospital admissions) are associated with daily changes in PM at very low background levels (say under 15 µg/m$^3$, and arguably up to 20 µg/m$^3$ or more), because of lack of studies of sufficient power to distinguish threshold and non-threshold models in these circumstances. This lack of information at very low concentrations is of practical relevance to applications in parts of Europe, notably Scandinavia, and to a lesser extent in the UK also. But given no clear evidence suggesting a threshold, and the fact that various previously suggested thresholds have all proved to be untenable, it seems prudent to consider that the relationship of PM to acute effects has no threshold at the population level.

(b) Other possible non-linearities in the shape of the relationship of PM with acute effects: This viewpoint regarding threshold is supported by wider considerations of the shape of the exposure-response relationship. Graphical representation of the London winter data suggests steeper effects, per unit of ambient smoke, when the background levels are low than when they are high; i.e., like the upper, rather than the lower half of an ‘S’ shape (Schwartz and Marcus, 1990). This suggests a linear form of relationship, after transformation to smoke on the log scale. In fact, Schwartz and Marcus (1990) found that a square root transform gave a better fit to the data. They also propose several plausible reasons for the observed shape.

In most other studies, the shape of the exposure-response relationship linking particles with acute mortality and morbidity is represented adequately by untransformed increases of PM, within Poisson regression analyses where expected deaths are represented in exponential (log-
linear) form. Several papers of Schwartz and co-workers present graphical evidence confirming this. Also, in these studies the exclusion of particularly high pollution days generally had no noticeable effect on estimated relative risks. (Non-linearity in the relationship might have shown as a higher RR in analyses with these data excluded).

The adequacy of untransformed indices of PM may be because the range of pollution studied as usually much smaller than in the London data and so there would need to be gross non-linearity before transformations would be required over these smaller ranges. Thus, in three recent studies, from Beijing (Xu et al., 1994), Athens (Touloumi et al., 1994) and Santiago (Ostro et al., in press), daily concentrations of particles are transformed to the log scale to represent a shape like that of the London data. The daily concentrations of particles in these three studies were very high.

There is also some evidence on non-linearity given by comparing results between studies. For example, Schwartz (1994a) lists several mortality studies which give quantitative estimates of a relationship between daily PM (expressed as TSP) and acute mortality. Comparison across studies suggests that the estimated effect, per 10 µg/m$^3$ difference in daily exposure, is negatively related with average background levels; i.e., a pattern similar to that found within the London, Beijing, Athens and Santiago studies.

In conclusion, although there is some evidence that relative risks per unit exposure are greater when background levels of PM pollution are lower, the simpler representation, of constant percentage increase in effect per unit exposure, without threshold, appears to be the best representation of the relationship under normal background conditions. Other relationships, estimated as non-linear (e.g., using logistic regression when studying presence/absence of symptoms) within epidemiological studies of interest, have been linearised for ease of application within the present project (See Section 4.6, below).

4.2.7 Who is at risk of acute effects on or following higher air pollution days?

Although the precise mechanisms of action are not well established, there is widespread agreement that those at risk of premature death or of increased, earlier or more severe hospital usage on higher pollution days are those with pre-existing serious ill-health. A mechanism has been conjectured whereby, in people whose health is already compromised by chronic disease and/or by ageing, the severity of an acute event (e.g., pneumonia, heart attack) may be increased on higher pollution days (Schwartz, 1994f). The issue of who is experiencing the acute effects has been investigated in several epidemiological studies.

(a) Mortality: Studies are limited by the lack of information readily available in most routine databases of mortality and hospital usage, and by the difficulties of gathering new relevant information. The most thorough recent study of who is dying on higher pollution days is of the Philadelphia data (Schwartz and Dockery, 1992b). Differential effects were found by age-group, the relative risk (RR) of mortality from non-accidental causes per 100 µg/m$^3$ TSP being 1.027 in those aged < 65 yr, compared with the much higher corresponding RR of 1.095 in those aged ³ 65; the overall RR being 1.068. Similarly, in Cincinnati, the RR per 100 µg/m$^3$ TSP overall was 1.06, whereas those aged 65 or more experienced the higher RR of 1.09 (Schwartz 1994b). The similarity of these results is striking. Early reports of excess mortality
from the London smog of 1952 also found higher risks among elderly people (Logan 1953), a point supported by further recent London data (Anderson et al, in press). In these studies, age may be a marker of increased likelihood of underlying chronic disease.

Considering the cause of death, Schwartz and Dockery (1992b) showed that the excess occurred for COPD (RR per 100 µg/m³ of 1.195, for pneumonia (RR 1.107) and for cardiovascular disease (RR 1.096), whereas the RR for cancer was substantially less than for all-cause mortality. Results from other studies where cause-specific mortality has been examined are consistent with these findings. For example, deaths from pneumonia and from cardiovascular causes were particularly elevated in Cincinnati (Schwartz 1994b); COPD and cardiovascular deaths were particularly elevated in Birmingham, Alabama (Schwartz 1993); the earlier Santa Clara study also highlighted respiratory and (to a lesser extent) circulatory causes (Fairley 1990) and cardio-respiratory causes were also found to be principally implicated in Athens.

Schwartz (1994f) reports further analyses of the Philadelphia data, contrasting those who died on high and on low pollution days (adjusted for weather, season etc). These analyses showed a substantial increase also in cases where respiratory disease, or vague respiratory cardiovascular symptoms, were reported as contributory causes to underlying cardiovascular mortality.

The limited evidence on who is at risk of excess mortality, therefore, supports the general viewpoints and conjectures reported above: what is involved is a shortening of life in people whose life expectancy was, in any case, much lower than average. (This is sometimes described as a ‘harvesting’ effect.) There is, however, little information about the distribution of length of life lost. However, the conjecture that the excess deaths would all have occurred in any case within a few days does not appear to be well-founded. For example, the same question of ‘harvesting’ has been investigated where excess deaths were weather-related (Kalkstein, 1993). There was some evidence of lower than expected deaths on days following excess weather-related mortality; but this subsequent drop accounted only for about 40% of the earlier weather-related excess. A mechanism like that suggested by Schwartz (1994f) seems more plausible. Serious events such as pneumonia or heart attack, against a background of already compromised health, will in many instances lead to death. The role, if any, of weather or pollution may then be to shorten life by at most a few days or weeks. In other instances, however, weather or pollution may precisely be what tips the balance between surviving this occasion or not; and the expected length of life lost may then be substantially greater than just a few days. Thus, the distribution of length of life lost is likely to be heavily skewed to the right, with a median of perhaps a few weeks, but a mean of perhaps several months. This is consistent with Schwartz’s (1994f) findings of a 22% increase in those dead-on-arrival at hospital on high relative to low pollution days, compared with an overall increase of only 5% in deaths generally. However, there is much conjecture in these remarks and reliable information is needed.

(b) Hospital usage: Studies of hospital admissions have focused on the effects of air pollution on daily admissions for respiratory conditions. Various other groups of conditions have been studied as control conditions and, as expected, have been found to be unrelated to
daily variations in pollution levels. The first results linking cardiovascular hospital admissions and particulate air pollution are now becoming available; but too late for inclusion in the present phase of this project. Within the broad framework of respiratory admissions, studies have variously looked at three broad groupings: (a) respiratory infections (or pneumonia); (b) asthma; and (c) other chronic obstructive pulmonary diseases (COPD). Evidence for an effect of PM pollution on respiratory infections and on COPD is stronger than on asthma (e.g. Schwartz, 1994d). It is unclear whether this difference is informative about possible mechanisms, or simply reflects greater power of studies to identify relationships with infections and COPD.

It is unclear to what extent, if any, the relative risks of admissions on or following higher pollution days vary by age-group within the population. In a large-scale study of hospital admissions for various respiratory conditions in Ontario, Canada 1983-88, similar pollution-related percentage increases were found across all three principal age-groups studied (2-34 y, 35-64 y and 65+), with an especially high percentage increase in the very young (0-1 y) (Burnett et al., 1994). These effects were attributable to ozone more than PM (measured as sulphates), though a relationship with sulphates was established also. Other recent papers however have examined ERVs (Schwartz et al, 1993) or hospital admissions (Schwartz 1994c; 1994d; 1994e; 1995) in the elderly (> 65yrs) only; presumably on the understanding that effects, if any, will be seen most clearly at these ages. This would be consistent with the mortality findings summarised earlier. It is unclear, however, what is the evidence for this age-specificity for hospital usage specifically.

(c) Restricted Activity Days (RADs): Design limitations of these studies have been noted earlier. In terms of results, it is noteworthy that relationships have been found not only with respiratory RADs (Ostro and Rothschild 1989), but with RADs irrespective of specific condition (Ostro 1987). Given the study limitations, it is unclear if this apparent non-specificity is real or artefactual. Note that the vast majority of RADs implicated were considered to be minor; i.e., not requiring staying in bed, or staying off work.

4.3 Acute Effects of Ozone

4.3.1 Introduction

(a) General remarks: Ozone is formed as a result of photochemical reactions, involving oxides of nitrogen and oxygen, which are catalysed by hydrocarbons. Levels of ozone rise during sunny weather and so there are substantial variations in concentrations both within a day and between days and seasons. The main sources of precursors of ozone are from motor vehicles and industrial processes; reduction in the precursors would lead to a reduction in ozone. Ozone is formed as the polluted air masses are transported away from their sources and so tends to be higher at a distance from the sources. Thus, concentrations are lower at sources close to NOx emissions such as in a busy street, rather than in rural areas, where there is a lack of this NOx sink. National levels are also affected by precursors of neighbouring countries’ ozone in the continent of Europe, and the actual mixture of pollutants will vary from place to place and from time to time.
(b) Toxicity: Animal Studies: The UK Department of Health Advisory Group on the Medical Aspects of Air Pollution Episodes recently reviewed the health effects of ambient ozone (MAAPE, 1991). Their review of animal studies showed that exposure to concentrations of ozone of the same order as experienced in the UK produced reversible histological changes in the terminal airways. It was agreed that the animal model was of more value in identification of the mechanism of effect and the pathological changes likely to occur, rather than as a basis for quantitative extrapolation from animals to humans. The animal studies also showed that exposure to ozone may produce tolerance to further exposure. The reasons for this ‘adaptation’ phenomenon are not clear. One possibility is that a damaged epithelium may not be capable of responding to further exposure. In addition, ozone has been suggested as a potential carcinogen and there is a need for further information in this area; although exposure to levels of ozone in the UK would be unlikely to lead to carcinogenic changes in the lung.

(c) Human Chamber Studies: Early assessments of the health effects of ozone in the companion US study drew heavily on the human chamber study results in providing quantitative risk estimates. In the present study we have used risk estimates based on epidemiological studies only, following a detailed review including many more recent studies (see later, this Section). This epidemiological focus better reflects the conditions under which incremental ozone from power stations is experienced by the population at risk. In particular, it avoids ‘translating’ from the relatively short-term exposures of chamber studies to daily ambient concentrations. Also, it is consistent with the approach for the other main pollutants studied.

Ozone chamber experiments suggest that increasing ozone concentrations cause respiratory symptoms and changes in lung function related to the duration of exposure and the concentration of ozone to which the individual is exposed. The effect is increased if exercising, especially if prolonged, as greater volume of air is inhaled with greater penetration of the lungs. The UK Department of Health report (MAAPE, 1991) concluded that changes in lung function are seen at concentrations of ozone which occur in the UK as described above but only in individuals undergoing moderate or heavy exercise for several hours. The dose-response curve tended to be non-linear with no threshold level for the effects of ozone on lung function.

The MAAPE report found no evidence that smokers, subjects with COPD, or the elderly were more sensitive to ozone than other subjects. However, only a small number of controlled chamber studies have been carried out on these aspects and these have tended to use small numbers of subjects. The evidence for asthmatics is less clear. There is some evidence that asthmatics may be more sensitive, at high ozone exposures, and that they may be more sensitive to SO₂ after exposure to ozone, but these studies were small and more work is needed on the effect of ozone in asthmatics. The MAAPE authors also suggest that individuals exposed to ozone concentrations in the UK were unlikely to sustain clinical symptoms except for subjects carrying out vigorous exercise for prolonged periods. However, if the exposure was superimposed on already impaired lung function, smaller changes could produce some clinical effects.
(d) Epidemiology: The principal epidemiological studies linking ambient concentrations of ozone with acute health effects have been carried out on the West Coast of the USA and in North-East USA/ South-East Canada. These two geographical locations have important differences in air pollution mixtures. For example, pollution in the US West Coast, notably California, is dominated by traffic. Air quality in NE USA/ SE Canada is strongly influenced by pollution from several major industrial US cities which gets carried across the Great Lakes (Thurston et al, 1994b).

Results of studies from these two locations, together with other recent studies, provide substantial epidemiological evidence of the acute health effects of ambient ozone. There was no established suitable review of this evidence when, during 1994, we considered what exposure-response functions, if any, to use within the present project. The UK MAAPE report, for example, does not attempt to quantify risks based on epidemiological studies. Consequently we reviewed the evidence afresh, by health endpoint. (That review has been partially updated in the light of papers published since mid-1994.)

4.3.2 Community-based studies of daily (acute) mortality

(a) Los Angeles County, California, USA

i. Kinney and Ozkaynak (1991) briefly review early analyses of daily mortality and air pollution in Los Angeles, 1956-58. Mills (1960) reported statistically significant associations between mortality and ozone in these data, adjusting for temperature and seasonality. Further and more comprehensive analyses by Hechter and Goldsmith (1961) also found positive associations, but these were not statistically significant. The authors concluded that an effect had not been demonstrated.

ii. Shumway et al (1988) considered daily mortality in Los Angeles County, CA, 1970-79, in relation to temperature, humidity and daily air pollution data averaged over six monitoring stations. The pollutants considered were CO, SO\textsubscript{2}, NO\textsubscript{2}, hydrocarbons (HC), O\textsubscript{3} and a measure of fine particles, some missing data being interpolated. In a separate examination of short- and long-term trends in the data, Shumway et al (1988) concluded that the principal associations with temperature and pollution were discernible in the longer-term trends. (This is in contrast with the body of evidence which focuses on short-term effects. Indeed, in many studies and locations the effective removal of longer-term trends is essential in order that an ozone effect be discernible. This is because ozone levels are highest in summer, whereas mortality and morbidity rates are usually highest in winter in locations studied.) Further detailed analysis considered the longer-term trends only. Not surprisingly therefore, no relationships with ozone were identified or reported.

iii. In further analyses, Kinney and Ozkaynak (1991) focused on short-term relationships. The effects of seasons and other longer-term trends were removed by a linear filtering of all variables, including mortality: each measurement was adjusted by subtracting from it a 19-day weighted moving average. This leaves intact the short-term, day-by-day variations. Pollutants considered were daily average SO\textsubscript{2}, NO\textsubscript{2} and fine particles; daily maximum 1-hr total oxidants (O\textsubscript{x}), averaged over eight monitoring stations; and daily maximum CO.
Annual average O\textsubscript{x} levels were fairly stable over the 10-year period, and averaged 75 ppb. It is unclear what was the contribution of O\textsubscript{x} specifically. However, corresponding O\textsubscript{x} concentrations 1981-89 averaged 72 ppb daily 1-hr max. (Kinney et al, 1994).

Within this framework, total mortality from internal causes was related principally to temperature, to O\textsubscript{x} (lagged one-day), and to any one of CO, NO\textsubscript{2} and particles. High short-term correlations between these latter three meant that their effects could not be distinguished individually. Further analyses of the filtered variables, examining NO\textsubscript{2}, temperature, and lagged O\textsubscript{x}, showed relationships with all three variables that were highly significant statistically, even though the variables jointly explained less than 4% of the total variability in adjusted daily mortality. The statistical modelling used ordinary linear regression methods; the (filtered) pollutant variables were untransformed (i.e., this was an additive, not a multiplicative model).

The effect of lagged O\textsubscript{x} was estimated as a slope of 0.030 (se. 0.008) per ppb O\textsubscript{x}. Given a study average of about 150 deaths per day from internal causes over the 10-year period, this is equivalent to an increase of 0.02% total mortality (se. 0.005%) per ppb increase in O\textsubscript{x}.

Although in principle the analysis gives equal weight to each day of the year, in practice day-by-day variations in O\textsubscript{x} levels were much smaller in the mid-winter season (when unfiltered O\textsubscript{x} levels were 20-50 ppb) than in summertime (when concentrations upwards of 100 ppb were not unusual). Thus, both high and low values of the filtered O\textsubscript{x} data came from summer days; the winter data had little influence on results. Indeed, analyses separately for the winter months (Dec-Feb) showed a statistically significant effect of temperature effect only. However, analyses restricted to the summer months only (June-Aug) again showed an effect of temperature, NO\textsubscript{2} and O\textsubscript{x}, the O\textsubscript{x} effect being very similar to that of the year overall.

Detailed sensitivity analyses showed that the relationship was insensitive to outliers or (individual) influential data points; to adjustment for systematic day-of-the-week effects or for autocorrelation; and to the filtering period, provided that it was 90 days or less. There was no evidence for non-linearity in the relationship. (Note that evidence for or against linearity is based principally on variation between days in summer and not on a summer-winter contrast.) Year-by-year analyses did show variations in the estimated coefficients; but all were positive, six were in the range 0.02 to 0.04, and there was no trend by year.

Analyses by cause-of-death showed that the greatest impact was on cardiovascular (CV) mortality (ICD 390-459): a statistically significant slope of 0.023 (se. 0.006) represents a 0.026% increase against the study average background of 87 CV deaths per day.

Only temperature was clearly related to deaths from respiratory disease (ICD 460-519). This may be due to small numbers: average eight deaths per day. The coefficient for O\textsubscript{x}, though small, was positive. Its high se. (0.012) relative to its estimated magnitude (0.002) imply that it may be misleading to interpret the estimated coefficient. Nevertheless, we note that
in percentage terms the O\textsubscript{3} effect on respiratory mortality is similar to the estimated O\textsubscript{x} effect on total or on CV mortality.

iv. Kinney \textit{et al} (1994: Abstract only) report a follow-up analysing daily mortality in Los Angeles County in subsequent years, 1981-90, in relation to a range of ambient air pollutants. These included O\textsubscript{3} and not O\textsubscript{x} as previously; though again expressed as the average over eight sites of daily 1-hr maxima. (O\textsubscript{3} data were available from 1979 only; mean value 1981-89 was 72 ppb.) Methods seem substantially the same as those of Kinney and Ozkaynak (1991); the abstract gives few details.

Using time-series and linear regression methods, and adjusting for season and other factors, previous-day O\textsubscript{3} was highly related statistically to daily mortality from internal causes. The estimated effect of 0.024 deaths/day/ppb O\textsubscript{3} (se. 0.008) was similar to that reported previously for total oxidants. In percentage terms, assuming as before about 150 deaths/day on average, this represents a 0.015% increase (se. 0.005%) in daily deaths. The slope was unchanged after exclusion of days when (previous-day) O\textsubscript{3} concentrations exceeded 120 ppb.

\textit{(b)} Santa Clara County, California, USA: In another Californian study, Fairley (1990) reported a relationship between particles and daily mortality in Santa Clara County, 1980-86. However other pollutants, including ozone, were not studied.

\textit{(c)} New York City: Kinney and Ozkaynak (1992: Abstract only available) analysed data from New York City 1971-76 using methods similar to Kinney and Ozkaynak (1991). Air pollutants considered were daily averages of SO\textsubscript{2} and coefficient of haze (COH) as an index of particulate air pollution; and, for April-September only, daily 1-hr maxima O\textsubscript{3} (mean O\textsubscript{3} 56 ppb for measurement period). Having adjusted for seasonality and studied lags, results were based on a linear regression model including temperature, relative humidity, COH and previous-day O\textsubscript{3}. The estimated slope for O\textsubscript{3} was 0.055 deaths/day/ppb; or a 0.034% increase in the overall number of daily deaths (mean 163 deaths/day; unclear if this was annual mean or summer months only).

\textit{(d)} Other North American studies: Many of the North American daily mortality studies focus on particles, SO\textsubscript{2} and NO\textsubscript{x}; ozone is not always included among the pollutants examined. However some studies, powerful enough to detect relationships of mortality with particles, have failed to find an effect of ozone: we give some examples.

Schwartz (1991), studying daily mortality 1973-82 in Detroit, uses average ozone data from typically 2-3 monitors throughout the city. (TSP and SO\textsubscript{2} data were based on monitoring at 10-15 sites). In a study using Poisson regression methods, adjusting for weather, temperature and season, and taking account of serial correlations, 1-hr. peak O\textsubscript{3} and 24-hr average O\textsubscript{3} were each ‘highly insignificant’ as predictors of mortality, including when winter months were excluded from the analysis. Average O\textsubscript{3} concentrations are not quoted. However, relationships with TSP were demonstrated clearly in this study.

Dockery \textit{et al} (1993) examined daily mortality over 12 months 1985-86 in St. Louis and in Kingston/ Harriman, Eastern Tennessee, using a similar strategy of statistical analysis. Air
pollution data, from a single monitor at each site, included hourly measurements of O₃. These exceeded the ambient standard of 120 ppb on five days in St. Louis, but on no days at Kingston. Statistical analyses used 24-hr averages (St. Louis mean 22.5 ppb, max. 64 ppb; Kingston mean 23 ppb, max. 49 ppb) rather than daily 1-hr maxima of O₃. The correlation between 1-hr daily max. and 24-hr average is not given. Adjusting for weather, temperature, seasonality etc., relationships of mortality with ozone were ‘far from statistical significance’ though various indices of particulate pollution were related to daily deaths.

(e) European studies: Ozone is also rarely if ever considered in the principal European studies of mortality, some of which (e.g., London) focus on winter pollution episodes. However, in an early European study of air pollution in Rotterdam, Biersteker and Evendijk (1976) found an apparent association between ambient O₃ and mortality, prior to adjusting for temperature and other effects. After suitable adjustment, however, the apparent relationship no longer obtained.

(f) Interim conclusions: Well-conducted studies of acute mortality have shown small but clear increases in the risks of daily deaths in two US locations: Los Angeles County (during each of two consecutive 10-year periods) and New York. These positive studies were in practice identifying an effect of summertime ozone pollution. In one of the three instances, exclusion of high-ozone days was examined and did not alter the estimated risks. However, some other well-conducted US studies have investigated and failed to find an ozone effect of acute mortality. There is little or no information from Europe specifically.

4.3.3 Hospital admissions

(a) (Southern) Ontario, Canada: Through the 1980s, Bates and Sizto published a series of papers examining hospital admissions for respiratory causes, notably asthma, at 79 acute care hospitals in a catchment area of about six million people in Southern Ontario, Canada. Studies during the 1990s, including by other researchers, have extended this important base of research findings. We summarise principal themes only, not all papers.

i. Bates and Sizto (1987) studied admissions for all respiratory causes, for asthma and for selected non-respiratory (‘control’) conditions for two winter months (Jan, Feb.) and two in summer (July, Aug) each year 1974-83 (1975 missing). Air pollution data from 17 monitoring stations gave hourly values of O₃, NO₂, SO₂ and coefficient of haze (COH), with 24-hour values of aerosol sulphates (SO₄) every sixth day. Levels of O₃ (mean daily 1-hr peak, averaged over 17 stations) were about 20-30 ppb and 50-70 ppb in the winter and summer months, respectively. Daily maximal 8-hr and maximal 1-hr average concentrations of O₃ were very highly correlated. Levels of SO₂ were higher in winter than in summer; other pollutants were similar in both periods. Bates and Sizto (1983) note heavy metal fabrication as one important source of particulate air pollution. The patterns of correlation variously between the pollutants, mean daily temperature and relative humidity were different in the summer from in winter months.

Daily numbers of admissions were adjusted by subtracting the average (mean) number of admissions for that same day of the week, in the same season, same year in order to
eliminate longer-term trends, and systematic day-of-the-week effects. Simple correlation coefficients suggested that in summer, air pollutants including ozone (unadjusted for one another, or for temperature and humidity) were related to general respiratory admissions and to asthma in particular; but not to non-respiratory admissions. In general, the winter admissions data showed relationships with temperature but not with pollution.

Multiple regression analyses were carried out. Results are reported very summarily and exposure-response relationships are not given. The authors were cautious about identifying any particular pollutant (e.g., O₃, SO₄) as causal because of the absence of relationships in the winter data. Rather, they speak of an ‘acid summer haze’ effect of the mixture as a whole; and suggest that unmeasured characteristics (e.g., H⁺) may be most relevant to causality.

ii. Bates and Sizto (1989) report limited regression results. These show SO₄ (24-hr lag) as the best single predictor among the pollutants studied. Inclusion of additional variables gave ambiguous results with regard to ozone, with positive coefficient for same-day (1-hr maximal) concentrations and a negative coefficient for O₃ lagged 48 hr when included together. These various indices of pollution are highly correlated individually and jointly, so that coefficients from such a complex model are difficult to interpret. The authors conjecture that neither O₃ or SO₄ is principally responsible, and again suggest H₂SO₄ aerosol (largely unmeasured in their study, and so not included in the regression analyses) as the main pollutant ‘driving’ the association with summer-time respiratory admissions.

iii. This difficulty of attributing effects to any one pollutant was illustrated in further analyses by Lipfert and Hammerstrom (1992) of data from Southern Ontario hospitals 1979-85. They concluded that a range of pollutants were plausibly related to admissions, especially July-August; and that it was not possible to attribute effects to any one pollutant in particular.

iv. Thurston et al (1994a, 1994b) investigated the acidity hypothesis in a study of admissions to Toronto hospitals, July and August 1986-88. Aspects of this study have been summarised earlier (4.2.5, above) where it was noted that, adjusting for non-pollution confounders, the clearest relationships with summertime respiratory admissions were found for ozone (Thurston et al, 1994b).

v. Burnett et al (1994) extended the research in the province of Ontario to include southern and central parts which contain 168 acute care hospitals covering 8.7 million people. Thus any observed adverse health outcome could be attributed to a large and diverse population and give increased power to detect health impacts. Ozone levels were obtained from 22 monitoring stations in the period of 1983 to 1988, while daily sulphate levels were recorded at nine monitoring stations. Ozone and sulphates on the day of admission and up to three days prior to the date of admission were both significantly related to hospital admissions for respiratory disease. Ozone was a stronger predictor of admissions than sulphates. The combined ozone-sulphate pollution mix was significantly associated with increased admissions for asthma, COPD, and infection. This was found in all age groups, with the greater impact in infants (15% of admissions associated with ozone-sulphate
pollution mix) and relatively little difference for other age groups. No relationship was identified with admissions in the winter months of December to March, when ozone levels and sulphate levels were low.

(b) New York State, North-East USA: Thurston et al (1992) studied the effects of summer (June-Aug) haze on respiratory hospital admissions, including asthma in particular, in four cities (New York City (NYC), Buffalo, Albany, White Plains) in the East Coast of the USA during the summers of 1988 and '89. The principal pollutants studied were H⁺, SO₄, O₃, and temperature. Temperature and O₃ were represented by daily maximum 1-hour values, whereas SO₄ and H⁺ were daily averages.

Average unscheduled hospital admissions varied little between 1988 and 1989. Respiratory admissions per 100,000 population were about 50% higher, and asthma admissions about 100% higher, in NYC than in the other three areas. Average and maximum temperatures were similar across the four study areas. On average, ozone concentrations were higher in Buffalo and White Plains than in Albany or NYC. The highest peak in ozone concentrations were in NYC in 1988, and maximum values tended to be higher in 1988 than in 1989. Average SO₄ and H⁺ concentrations were reasonably similar across the areas and years, with a high 'spike' for H⁺ in August 1988.

Analyses adjusted for day of the week effects, and took out time trends by using filtering. Ordinary least squares regression models of the filtered admissions data adjusted in addition for temperature and examined the effects of each of the pollutants in turn (but not simultaneously). The temperature and pollutant variables were lagged by up to 3 days, depending on which lag gave the highest significance. Normality and autocorrelation tests were carried out on the residuals from the regressions.

Results varied by location and by year. Graphs showed that relationships were stronger with general respiratory admissions than with asthma admissions. At all four locations, the evidence for a relationship was stronger in 1988 than in 1989. For both years, the evidence for a relationship was stronger for Buffalo and NYC than for Albany and for White Plains.

Detailed regression analyses of filtered admissions data on temperature and one pollutant were restricted to Buffalo and NYC in 1988 only. For respiratory admissions (which include asthma) the equation with temperature and H⁺ gave the highest correlation for both Buffalo and NYC, with an ozone effect also indicated in NYC. For asthma admissions, the equation with temperature and ozone gave the highest correlation in Buffalo, but the equations with H⁺ and SO₄ fitted the data better in NYC. However, the estimated ozone effect was the same in both locations.

Thus, this study gives evidence linking summertime air pollution in New York State with ERVs for respiratory causes, and for asthma in particular. However, the evidence of an effect varies by location and year. This raises serious doubts about the usability/ transferability of the positive relationships, across space and/or time. Moreover, where statistically significant positive associations were identified, it was not possible to describe clearly the relative roles of O₃, of SO₄ and of H⁺; or indeed of unmeasured indices of air pollution. The discussion talks about a possible combination effect of ozone and H⁺, the highest ozone effects being found
when acidity $(H^+)$ was also high. However, no regression results are given to support this view.

(c) *Other key US studies:* In four studies in different locations Schwartz considered the effect of ozone and particles on daily hospital admissions for respiratory disease in the elderly, i.e., those aged 65 years or more (Schwartz 1994c, 1994d, 1994e, 1995).

i. Minneapolis-St. Paul, Minnesota: Daily admissions for pneumonia (ICD9 480-487) and COPD (ICD9 490-496) were obtained for the period 1986 to 1989, in those aged 65 years and older (Schwartz, 1994e). Ozone was measured hourly and the average for each monitor over 24 hours was then averaged over all monitors. Also the daily maximum 1-hour reading was recorded. Poisson regression was used to model daily admissions. Dummy variables were used for temperature and dew point in the regression analysis. Alternatives such as generalised additive models and cubic splines were also used to fit the non-linear dependence on time and weather. Controlling for time trends, seasonal fluctuations, weather and PM$_{10}$, ozone was associated with admissions for pneumonia (RR = 1.15, 95% CI = 0.97, 1.36 for increase in 50 ppb in daily (24-hr) ozone concentration) but not for COPD. Without adjusting for PM$_{10}$, the relative risks were similar but statistically significant (RR = 1.19, 95% CI = 1.02, 1.40 for 50 ppb increase in ozone). Results were also insensitive to several other variations in the model or data. However, the use of peak 1-hr rather than daily 24-hour average ozone values markedly reduced the statistical significance of the association.

ii. Birmingham, Alabama: In a similar study, daily admissions for COPD and pneumonia were obtained for the period 1986 to 1989 (Schwartz, 1994c). As in the previous study, ozone was assessed using average 24-hour and average daily maximum 1 hour readings. A Poisson regression model was used to assess daily admissions for persons aged 65 years and older, controlling for time trends, seasonal fluctuations and weather. Same-day ozone was not significantly associated with hospital admissions. However, ozone was usually associated with admissions for pneumonia with a 2-day lag and for COPD with a 1-day lag. The results were not sensitive to the methods for controlling for seasonal patterns and weather, nor to the exclusion of very hot or cold days.

iii. Detroit: In a third study, hospital admissions were studied in Detroit, in the period 1986-1989 (Schwartz, 1994d). Ozone was measured as in the previous two studies. A Poisson regression model taking account of over-dispersion was used, and controlling for seasonal and other long-term time trends, for temperature and dew point temperature. Adjusting for PM$_{10}$ levels, 24 hour ozone concentration was significantly associated with daily admissions for COPD and pneumonia. The magnitude of the risks were similar to those reported in Birmingham, Alabama.

iv. New Haven, Connecticut and Tacoma, Washington: The fourth study of the elderly (Schwartz, 1995) looked at the relationship of short term changes in air pollution and hospital admissions for respiratory disease (ICD9 460-519) in these two cities, which had similar levels of particles (PM$_{10}$) but where concentrations of SO$_2$ and of ozone were much higher in New Haven. Each city was analysed separately and daily counts of admissions
were constructed for individuals aged 65 years and over. Ozone was measured hourly, and the 24 hour average of each monitor was averaged over all monitors for seven months of the year (April - October), as ozone was very low during the cold months. In the regression analysis using a Poisson model, a 19-day weighted moving average regression filter was used to remove all seasonal and sub-seasonal patterns from the data. A potential U-shaped dependence of admissions on temperature was dealt with by using eight categories of temperature and humidity. The daily admissions were initially regressed on temperature, humidity and day of the week indicators, followed by pollution variables, individually and then in pairs. In Hew Haven increased daily admissions were associated with ozone, adjusting for PM$_{10}$ (RR = 1.07, 95% CI = 1.0, 1.15 for 50 µg/m$^3$ increase in ozone), while in Tacoma the ozone-related effect was higher (RR = 1.20, 95% CI = 1.06, 1.37 for 50 µg/m$^3$ increase in ozone.) These relative risks associated with ozone were similar to those in regression models unadjusted for other pollutants, suggesting an independent effect of ozone on daily respiratory admissions.

(d) European studies: There is little epidemiological information from Europe, partly because ozone concentrations are in general lower than in the USA. This also increases the difficulty in identifying a positive effect, if one exists, in those studies that are conducted. Thus, Ponka and Virtanen (1994) included ozone in their study of hospital admissions for chronic bronchitis and emphysema in Helsinki, 1987-1989. The mean daily concentration of O$_3$ was described as ‘fairly low’, at 11 ppb. At these concentrations, there was no observed association with admissions.

Results from Europe are not uniformly negative, however. Specifically, analyses within the EC-sponsored APHEA project of London hospital respiratory admissions data identified ozone as the pollutant most clearly associated with daily admissions (Ross Anderson, personal communication). These and other European results will be published in due course.

4.3.4 Emergency room visits (ERVs)

The studies reviewed gave mixed results.

(a) Negative results: Several failed to find an association between ozone and ERVs. Thus, Richards et al (1981) did not show a relationship between ozone levels in Los Angeles and emergency visits or admissions at a children’s hospital. Similarly, Samet et al (1981), studying ERVs for respiratory diseases in Steubenville, Ohio during selected months 1974-77 found relationships with TSP and/or SO$_2$, but not with ozone.

Sunyer et al (1991) studied emergency room admissions for COPD in Barcelona, 1985-86. Ozone data (daily 1-hr maxima, averaged over two stations), reported as µg/m$^3$, showed annual mean 63, sd 43, maximum 253 (98th percentile 182). Unadjusted data showed a negative association between asthma admissions and ozone concentrations, whereas crude associations were positive for the several other pollutants considered. (Such negative unadjusted associations with O$_3$ are not unusual, because of seasonal effects.) Adjusting for meteorology, season and day-of-the-week showed statistically significant relationships with SO$_2$, black smoke and CO, but not with O$_3$ or NO$_2$. There was no evidence of synergism
between ozone and either SO$_2$ or black smoke. Separate analyses were carried out by season; but it seems that ozone was not examined in these.

Schwartz et al (1993) studied air pollution and emergency room visits for asthma at eight hospitals in Seattle, Sept. 1989 through Sept. 1990. As in Vancouver, visits for asthma peaked in September of both years. Ozone data, from a site 20 km east of Seattle, were available May through September only. Using Poisson regression analyses, taking account of autoregression and extra-Poisson variability and adjusting for temperature and season, ozone was unrelated to attendance for asthma (the estimated association was marginally negative) whereas a clear relationship with particles was found. Otherwise, information on O$_3$ is sparse: it is not stated what index (e.g. daily 1-hr peak, daily average) was used; what were the background levels; or what lagging was investigated.

(b) Ambiguous results: Bates et al (1990) examined attendance at emergency departments over 28 months during 1984-86 at all (nine) acute care hospitals in Vancouver, Canada. They reported on average about 25,500 visits per month, including 2.7% for respiratory conditions, of which 41.3% were for asthma (ICD 493). Special efforts were made to ensure consistency of diagnosis. Air pollution data from 11 monitoring stations were represented as mean daily 1-hr maxima of COH, SO$_2$, NO$_2$ and O$_3$ (mean value about 20 or 30 ppb in summer and winter, respectively). Limited data on SO$_4$ suggested levels much lower than in Southern Ontario; with little or no acid aerosols in Vancouver. There was a strong correlation between O$_3$ and temperature, both in summer and in winter, as well as other inter-correlations among pollutants and temperature.

The study was intended as exploring feasibility and generating hypotheses. A strong September peak in asthma admissions was unexplained by the pollutants measured. Limited analyses, unadjusted for temperature, suggested an association between O$_3$ and total hospital visits, almost certainly reflecting a temperature effect. There was some evidence that summer visits for respiratory conditions were related to SO$_2$ or SO$_4$, but not to O$_3$. Bates et al (1990) interpret this as supporting evidence that the Southern Ontario hospital admissions results may reflect confounding with sulphuric acid aerosol, rather than a relationship with O$_3$ as such. They emphasise the difficulties of identifying reliably the effects of SO$_2$, NO$_2$ and O$_3$.

(c) Positive findings:

i. Cody et al (1992) studied emergency department visits for asthma and bronchitis in New Jersey for the summer (May-Aug) of 1988 and 1989. Ozone, SO$_2$, temperature and visibility data were collected hourly, and PM$_{10}$ was measured every 6th day, daily ozone concentrations being expressed as 5-hour averages (10.00-15.00). Ozone was positively correlated with temperature and SO$_2$, negatively correlated with humidity and visibility (excluding rain days), and apparently unrelated to bronchitis visits.

Temperature had a highly statistically significant association with asthma visits. (The coefficient was negative: there were fewer ERVs for asthma on hotter days.) Regression modelling, adjusting for temperature (but not for particles: the sparse PM$_{10}$ data seemed unrelated to asthma ERVs) showed statistically significant effects of ozone on emergency asthma visits in each year separately and in the years combined.
There are important and unexplained differences between years in the estimated ozone effects, possibly due to incomplete adjustment for other factors. (Average ozone levels were higher, and average ERVs for asthma were lower, in 1988 than in 1989). This may affect transferability of results. Further details of this study, especially results, are given later.

ii. White et al (1994) studied the relationship between asthma in mainly black 1-16 year old children and the number of emergency clinic visits. They found a significant positive relationship between the number of emergency visits and ozone concentrations above 0.11 ppm on the previous day, adjusted for minimum temperature, average PM_{10} level on the previous day and day of the week. Among low socio-economic families, asthma may be exacerbated following periods of high ozone concentration. This result is difficult to apply within an E-R framework because a continuous measure of ozone was not used.

### 4.3.5 Restricted activity days (RADs)

(a) **Adults: The US Health Interview Study (HIS):** Several papers have studied air pollution and restricted activity days (RADs) using data on adults (i.e., aged 18 or more) from the US Health Interview Study (HIS). These are studies of acute effects, relating the number of RADs in a two-week period just prior to interview with pollutant measures in concurrent or earlier two-week periods. However, in contrast to the other acute effects studies included in the present project, these various RAD studies based on the HIS are cross-sectional rather than longitudinal in design. Thus, they suffer from the possible limitation of inadequate adjustment in the analysis for subject-specific confounders which are controlled by design in well-conducted panel studies.

In the nomenclature of these papers, a RAD is any day where the subject was forced to alter his or her normal activity and includes bed disability, work loss, and other minor restrictions. A RRAD is any RAD caused by an acute respiratory condition. A MRAD is a minor restriction i.e., neither bed disability nor work loss.

Portney and Mullahy (1986), using HIS data from 1979 only, reported relationships with ozone, principally. Ostro (1987) carried out separate analyses of HIS data from the six years 1976-81. He reported relationships with fine particles (PM_{2.5}) but not with ozone. Ostro and Rothschild (1989) re-analysed the 1976-81 data, examining both fine particles (FP) and ozone but restricting the population studied to adults aged 18-65 who are currently working, and resident in urban areas; and again found relationships with both FP and ozone. Further details of these studies are given later.

While these are important results, given the scarcity of studies of RAD as an endpoint, there are also important design limitations in the HIS studies. These include:

i. The cross-sectional design, and hence importance of correctly and fully adjusting for confounders. The year-by-year differences in results, together with the differences between different studies of HIS data, may reflect inadequacy of the basic regression model. The high correlation between ozone and temperature as measured is a particular concern here.
ii. Recall bias, especially when the respondent was a family member, not the study subject.

iii. Possible inappropriateness of 2-week average (of 1-hr daily max.) ozone as a pollution index explaining the number of RADs in the concurrent two-week period.

iv. Selection effects in defining the study group.

(b) Children: School absences: Mexico City: In a study in Mexico city, Romieu et al (1992) studied respiratory-related ‘school’ absenteeism (at least two consecutive days absent due to respiratory illness; presumably as notified by parents) Jan-Mar 1990 among 111 children at a private kindergarten. As well as O$_3$ and temperature throughout the period, data on NO$_2$ and PM$_{10}$ were available for part of the study period. The correlation between temperature and O$_3$ (daily 1-hr peak) was very low. The O$_3$ concentrations exceeded 200 ppb in about one-third of the days; and exceeded 110 ppb on 85% of days; i.e., these were values much higher than would apply generally in Europe. Mean daily average PM$_{10}$ was 40 µg/m$^3$ on days measured.

Logistic regression methods, taking account of autocorrelation, were used to examine O$_3$ (daily 1-hr peak, unlagged and lagged; cumulative, defined as 1-hr peak exceeding 130 ppb on at least two consecutive days) in relation to respiratory-related absences, adjusting for passive smoking and temperature (but not other pollutants). Ozone, treated as a categorical variable, was significantly related to absenteeism. A ‘cumulative’ measure of ozone fit the data best, reflecting results from another study in Mexico City (Castillejos et al, 1992: see 4.3.6, below). The paper does not report analyses of O$_3$ on a continuous scale; hence, no conventional exposure-response results are given.

4.3.6 Asthma attacks

In panel studies, the (respiratory) health of individual subjects is monitored over a number of days, weeks or months. Changes in health (lung function changes; prevalence, incidence or duration of symptoms or symptom episodes) over time are then examined in relation to changes in ambient pollution, taking account where possible of other time-dependent covariates (temperature, humidity) or risk factors, usually at the individual level.

Exposure-response relationships linking asthma attacks with total oxidant or ozone exposure can be obtained from two panel studies of asthmatics, by Whittemore and Korn (1980) and by Holguin et al (1985). Both studies provide good evidence that a real association exists, but both are flawed from the viewpoint of providing usable exposure-response relationships.

(a) Los Angeles: Whittemore and Korn (1980): Whittemore and Korn (1980) studied 16 panels of asthmatic subjects in the Los Angeles area of the USA. This is a seminal paper for its methods of statistical analysis. However, there was a very high dropout rate, of about 50% of subjects. Also, the subjects studied were relatively severe asthmatics and so are probably untypical of asthmatics generally. Air pollutants were highly inter-correlated, complicating the task of identifying an effect of oxidants specifically. Finally, relationships are with total oxidants rather than with ozone; and refer to 24-hr average values, not 1-hr maxima.

Asthma attacks were significantly related to oxidants and TSP, adjusting for whether an individual had an attack the previous day, for minimum temperature, relative humidity, and day of the week. Given the limitations discussed above and assuming a background
probability of 0.1, these results correspond to an increase of 9 attacks per day per 1000 asthmatics at risk for a 100 mcg/m$^3$ increase in TSP and an increase of 20 attacks per day per 1000 asthmatics at risk for an increase in 0.1 ppm of oxidants.

(b) Houston, Texas: Holguin et al (1985): Holguin et al (1985) is a summertime (May-October, 1981) study of young (median age 13 years) non-smoking asthmatics near Houston, Texas, USA, where ‘the composition of summertime photochemical pollution may be quite different from other well-studied areas’. It appears that those studied were asthmatics of moderate severity. Results, based on 42 subjects only, refer to 12-hour periods (‘daytime’, 7am-7pm; and ‘night-time’, 7pm to 7am). The attack rate of asthmatic episodes, defined by occurrence of symptoms, decreasing peak expiratory flow (PEF) and increased use of medication, was 15%. The definition varied by subject. Results are not reported separately for day- and night-time.

Ozone and NO$_2$ were studied. Each was represented by the maximum 1-hr concentration in the relevant 12-hr period. These are estimates of individual exposures, taking account not only of ambient local concentrations, but also of individual activities such as time spent indoors and in vehicles. The relationship between these personal estimates and measured ambient O$_3$ concentrations is not given. Daytime NO$_2$ and O$_3$ were positively correlated: extent not given. Particles were measured but no results reported: reason not given.

Subject-specific logistic regression analyses included attack status in the previous 12-hour period, temperature, relative humidity, pollen count and NO$_2$. In this context, O$_3$ was positively related to asthma attacks for most subjects. Interactions (e.g., of O$_3$ with previous attack) were not examined; nor were possible dependencies on attack status earlier than previous period. Lagged exposure variables were not included. Further details are given later.

4.3.7 Respiratory symptoms and lung function: children

Many panel studies focus on lung function as the health effect of choice. We consider lung function only briefly, to provide context, because at present it cannot be valued monetarily.

(a) Children; summer camp studies: Good reviews of panel studies are provided by the UK Advisory Group on the Medical Effects of Air Pollution (MAAPE, 1991). Many of the studies considered were from North America, were conducted in summer camps, and studied lung function as an endpoint. (Both the pattern of exposure and the exercising habits of children in summer camps are likely to differ from those in home neighbourhoods.) Summarising the main results, MAAPE (1991) reports that these studies consistently show a negative association between ozone and lung function, which is probably causal. Both FVC and FEV$_1$ are affected, indicating a restrictive effect principally. Some individuals appeared to respond severely even when average effects were small. The effects are greater than would be predicted from exposure chamber studies, possibly reflecting longer exposure times, or inadequate adjustment for other pollutants, or real synergism between ozone and other pollutants. Pollution in the home neighbourhoods may have caused adaptation, so underestimating the effects in panel studies. It is unclear whether or not a threshold exists
Public Health Effects

(effects have been found at levels below 80 ppb); any threshold is likely to depend on individuals’ activity patterns as well as on ambient concentrations.

MAAPE (1991) considers symptoms only briefly but concludes that in these studies, ‘Where symptoms have been measured there has generally been no association between symptoms and ambient ozone level’ (p73).

(b) Children: home neighbourhood studies: Many studies do not examine ozone as a pollutant of interest. Thus, for example, the well-known panel studies by Pope et al (1991) and by Pope and Dockery (1992) in Utah County, Utah, did not examine ozone effects; probably because these were winter-time studies with consequently low ozone concentrations. There are nevertheless a number of studies which report positive findings.

i. Castillejos et al (1992) studied lung function and acute respiratory symptoms in 143 7- to 9-yr-old children from three private schools in Mexico City. For each participant, health effects were studied fortnightly, on about 10 occasions January to June 1988. Ambient pollution was high, due to traffic, industry (Romieu et al, 1992) and the topography of the area (Castillejos et al, 1992). Data, available for O₃ only, showed 1-hr peak concentrations in excess of 120 ppb on 74% of the days. Ozone concentrations at times of medical surveys were correlated positively with ambient temperature, and negatively with relative humidity.

Logistic regression analyses, adjusting for epidemics and individuals’ susceptibility to symptoms (but apparently not adjusting for temperature and humidity) showed that mean O₃ in the previous 48-hr was associated with children’s reports of cough or phlegm. (The apparent association of O₃ with lung function was reduced following adjustment for temperature and humidity.) The relative odds for an increase of 53ppb O₃ was 1.88 (95%CI 1.07, 3.30). (The average rates of reporting cough and phlegm are not given.) Shorter-term indices of O₃ concentrations were unrelated to symptoms, suggesting tolerance or adaptation. (When background levels of O₃ are very high, short-term peak concentrations may be less relevant physiologically (Lippmann, 1989).)

The authors are very cautious in interpreting the positive symptoms finding, because it was not reflected in the other O₃ analysis, and because the children’s self-reports of symptoms may have been unreliable. Note that this study, like the other Mexico City study by Romieu et al (1992) above, suggests an effect of ozone on the health of school-age children. However neither study gives usable exposure-response relationships. Also, the very high ambient O₃ levels create a distinctive situation; including, for example, that cumulative rather than immediate O₃ may be the relevant exposure index.

ii. Schwartz et al (1994) studied the association between ambient air pollution and respiratory illness in approximately 300 elementary school children. A daily diary was kept of each child’s respiratory symptoms for one year, while daily ozone levels were calculated from the hourly measurements. The analyses were limited to the five warm season months between April and August. Logistic regression of the respiratory symptoms was used with allowance for autocorrelation using the approach of generalised estimating equations (GEE). The form of the relationship of pollutants to the probability of symptoms was
assessed using generalised additive modelling (GAM). A change in 24 hour ozone concentration of 30 ppb was associated with a relative odds of 1.22 (95% CI = 0.96, 1.49) for the incidence of coughing. This relationship was independent of other pollutants.

iii. Hoek et al (1993) reported relationships linking previous-day peak ozone with a range of lung function measurements in a study of 533 school-children aged 7-12y, in the Netherlands. Although not directly of use in the present context, this study shows that ozone effects on the lung may be identifiable in panel studies in Europe also; and hence supports the transferability of positive US findings.

4.3.8 Respiratory symptoms: adults

Two series of studies of adults, both carried out in California, USA, provide evidence that adults too may experience more symptoms following days of higher ozone pollution. The first series is based on a symptom diary study of student nurses in Los Angeles, 1961-64. Recent re-analyses of these data showed statistically significant relationships between total oxidants and the nurses’ reports of coughing and eye irritation (Schwartz et al, 1988). Further analyses, focusing on the duration of symptom episodes, showed highly significant relationships statistically between daily photochemical oxidants and coughing, phlegm and sore throat. While strongly suggestive of an effect, these studies did not control for particulate pollution. Also, their use of total oxidants rather than ozone specifically is a limitation for present purposes.

Analyses of another major panel study in California also found strong relationships linking daily ambient ozone with presence of symptoms (mostly respiratory) in adults (Krupnick et al, 1990; Ostro et al, 1993). These findings are considered in much greater detail later.

4.3.9 Discussion/conclusions/strategic judgements/risk model assumptions

(a) Acute health effects of ozone: summary overview of the evidence.

- Although there is a substantial body of epidemiological evidence on the acute health effects of ozone (positive or negative findings), many studies do not include ozone as an explanatory variable. This may be because suitable data are unavailable. Or, it may reflect a prior belief that ozone effects are unidentifiable in that time or place (e.g., because daily ozone variability is too low for effects to show).

- Nevertheless many well-conducted epidemiological studies have shown associations between daily concentrations of ozone and adverse health effects. The general thrust of these studies is of effects lagged one or two days. There are also many well-conducted studies which do not show an acute ozone effect.

- For endpoints such as symptoms or lung function changes, these positive results are qualitatively consistent with, though often smaller in magnitude than, the effects which have been well-established in human chamber studies. The chamber studies show increased effects among people during (heavy) exercise. There is also positive evidence from animal experimental studies.
• The observed acute effects in epidemiological studies have been found over a very wide range of health endpoints, from mortality through hospital usage (respiratory causes) and asthma attacks to symptoms and lung function.

• Following the tradition of chamber studies, many epidemiological studies have used 1-hr daily maximum ozone as the index of choice. Increasingly however the ozone concentrations over longer periods within a day (5hrs, 8hrs, 24hrs) are being used. This reflects a view that exposure over these longer time-periods may be more relevant biologically. *Within a location* these various indices tend to be (very) highly correlated. The conversion factors will however vary *between locations* and so the choice of index is not irrelevant.

• The positive epidemiological evidence is principally but not exclusively from summertime data, when ozone levels are highest. Many but by no means all of the positive studies have been carried out in three regions of North America where ozone concentrations are relatively or very high: California, Ontario/ North-East USA and Mexico City. Nevertheless, some epidemiological studies conducted elsewhere in the US have reported positive findings, especially with hospital usage endpoints. There is also some positive evidence from Europe.

• Some studies exclude the highest-ozone days and still find mostly unaltered E-R relationships; i.e. this is not solely an episodic, very high ozone, issue. Rather, at high levels of ozone (e.g. Mexico city) acute effects may be reduced per unit exposure, possibly indicating adaptation; and some measure of ‘recent cumulative’ exposure may be most relevant.

• There is no strong or consistent evidence *within studies* of a threshold in the acute effects of ozone. Many studies do not investigate the issue explicitly; i.e. either the issue has been overlooked in analyses, or no evidence was found and the observation not reported. There is some evidence, from some studies, of no increase in risk at low daily concentrations.

• There is substantial variation in the general air pollution mixtures within which positive ozone effects have been found. In Ontario/ NE USA, it has proved difficult to disentangle ozone effects specifically from the wider summertime haze pollution mixture of which it forms a central part. There may be complex synergisms between pollutants. This may explain why estimated effects in Ontario/ NE USA studies can vary substantially by location and/or by year.

• Nevertheless, the general thrust of the epidemiological evidence is of an ozone effect which can for practical purposes be considered as independent of and additive to the effect of particles. (The correlation coefficient between daily particles and ozone is usually not high. Regression coefficients for one pollutant are usually insensitive to inclusion or not of the other.)
(b) Strategic judgements about causality and transferability

There appears to be little doubt that in some locations and circumstances, daily ozone concentrations are related to various acute health endpoints in ways that are quantifiable. The main problem, which is a practical as well as a theoretical one, is why effects are rarely detected except in relatively high-ozone situations (given that in these situations, there is not consistent or strong evidence of threshold levels). Is this because only high levels have an effect? Or, because day-to-day variations in O₃ concentrations at low levels are so small that an effect, though real, is undetectable? Or are there other explanations?

Against this background, possible interpretations include:

i. Ozone is a primary causal agent which always can cause problems; but these are detectable only with high levels/ high variability (e.g., summertime studies in some locations). However, increased ozone concentrations will have some, possibly unmeasurably small, effect of increasing risks in other situations also.

ii. Ozone causes problems only under the associated circumstances of high daily concentrations and high daily variability; i.e., where concentrations exceed some threshold and vary sufficiently so that mechanisms of adaptation do not seriously inhibit effects. Under this scenario, there would in reality be no effect on risks under other circumstances.

iii. An elaboration of this view might characterise those most at risk under such circumstances; e.g., those who, by exercising heavily outdoors, experience a relatively high dose of ozone.

iv. Ozone is not really the problem at all; but high O₃ concentrations (and associated high daily variability in O₃) are indicators of climate conditions and associated pollution mixtures which do cause problems; and where the real causal agents have not been well (enough) measured or represented in the epidemiological studies, or the studies (maybe due to lack of power) have not examined interactions well. A slightly softer version would include some role for ozone as part of such a mixture, possibly in interaction with other pollutants or climate; but it would be wrong to ascribe the health effects of the mixture to O₃ as such.

Evidence for the third and most sceptical of these interpretations comes principally from Ontario/ NE USA; and even here the most recent studies (e.g., Burnett et al, 1994) have identified ozone effects quite clearly. While this viewpoint may be tenable, we think that the thrust of evidence elsewhere, and so of the evidence as a whole, favours one of the earlier two interpretations. This is consistent with a view that at least in the locations studied the identified relationships with ozone are causal and so (with suitable qualifications) transferable to other sufficiently similar circumstances; and that these quantified ozone effects can be considered additive to those of particles from the viewpoint of estimating the joint effects of both pollutants.
The question then is how widely to generalise. Here, on the evidence, we favour the first interpretation: that effects are best represented by considering ozone as a causal agent more-or-less independent of circumstances; but noting that effects do in practice vary with circumstances more than arises for particle effects. This is consistent in practice with an application of E-R relationships to incremental ozone irrespective of background levels or other circumstances; i.e., the same approach as for particles.

There are problems in modelling the incremental effect on ambient ozone concentrations of the emissions of ozone precursors from high stacks. Within the ExternE Project our analysis, demonstrated so far only for the German implementation of the coal fuel cycle, is based solely on increments for high O₃ days. There would be substantial uncertainties in extrapolating the ozone modelling results to low concentration days also. Consequently, estimation of the acute health effects of incremental ozone has been restricted to days when background O₃ is higher than 60 ppb. This is not to be interpreted as no adverse health effects at ambient O₃ concentrations below 60 ppb. As noted above, the evidence from epidemiological studies is unclear on what if any is a threshold for the acute effects of ambient ozone.

4.4 Chronic Effects

4.4.1 Introduction

The methodological problems in assessing the relationship between air pollution and chronic health impacts raises more difficulties than in acute studies. One major difficulty lies in determining biologically relevant estimates of exposure for the individuals studied. To some extent this reflects a lack of underlying knowledge about disease mechanisms. For example, what pollutants may be involved? And, for a given pollutant (say particles measured as PM₁₀), what is the biologically relevant period of exposure (e.g., childhood; recent years; cumulative lifetime)? In principle, exposures may need to be estimated over the full lifetimes of the individuals studied. The difficulties in exposure estimation also reflect practical constraints, with lack of information about past concentrations of pollutants in the various (micro)-environments where study subjects have spent time; lack of information about individuals’ time-activity patterns relevant to exposures in these locations; and lack of exposure estimation methodology for estimating these components of exposure. There is a clear need for methodological work in this area.

In the face of these difficulties, the biologically relevant personal exposures of individuals to ambient pollutants are usually represented or approximated by recent ambient concentrations from regional (fixed-point) monitors. Clearly, this is at best a crude approximation; and both systematic and random errors in the estimated exposures of individuals (relative to their true, biologically relevant exposures) weaken the power of studies to identify real relationships and may distort risk estimates from those relationships which are identified.

The second major difficulty is in adjustment for confounding factors; i.e., those differences between individuals which in practice may be correlated with exposure to ambient pollution but are not because of it, and which are also related to the development of chronic disease. Key examples include present and past socio-economic status; lifetime smoking habits; and
(affordable) access to medical services. The role of illness is also relevant; though its status as a possible outcome of earlier exposure, as well as a predictor of later disease and death, means that care needs to be taken in whether or not to adjust for illness in assessing exposure effects.

Note that the possible confounding effects of these factors were more-or-less eliminated in the studies reviewed in sections 4.2 and 4.3, both the time-series studies at the population level and panel studies of individuals. This is because the contrasts of primary interest in these studies were changes over time in health, pollution and other factors, in study populations whose socio-economic and lifestyle characteristics over time were more-or-less unchanged. Most studies of chronic effects are however cross-sectional in design, involving contrasts between study subgroups (often defined by area of residence) at a point in time, rather than in the same group over time. Hence the need to take account of confounding factors that vary between individuals.

The development of statistical methods to overcome these difficulties has been identified as a priority (Prentice and Thomas, 1993). Meantime, the problems have two principal implications for our purposes. First, whereas there have been many epidemiological studies investigating the chronic health effects of air pollution, methodological limitations imply that many of these are of limited reliability even in terms of their qualitative conclusions. Secondly, the confidence that can be placed in the quantitative risk estimates even of the better studies is less than that for acute effects. Consequently, the estimation of chronic health effects was not attempted in the early phases of the ExternE project. This, despite the fact that the development of chronic disease is, for an individual, more serious than many of the acute effects that had been estimated; an understanding that might reasonably be reflected in economic valuation.

The position changed late in 1993 with the publication of a major cohort study of the chronic mortality effects of air pollution (Dockery et al, 1993) and knowledge that another major cohort study (later published as Pope et al, 1995c) was in progress. These cohort studies took account of confounding factors at the individual level. This methodological advance, together with the importance of the endpoint, encouraged us at least to make a start towards estimating chronic disease effects.

Since then, a number of other major reviews have been published (Lipfert, 1994; Pope et al, 1995a; Pope et al, 1995b). These reviews supplement the following brief write-up, which focuses principally on papers which were considered in mid-1994. (We will re-consider the issue of E-R relationships for chronic effects if, as we hope, ExternE moves into a further phase.) In particular, just as in mortality studies the cohort design is an improvement of cross-sectional studies of death rates, so in morbidity studies we wish to base risk estimates where practicable on longitudinal (repeated measures) designs rather than on cross-sectional studies. For example Ostro, developing risk estimation methods within the Hagler/Bailly State of New York study (Rowe et al, 1994), identified longitudinal relationships by Abbey et al (1993) which seem more suitable than those used here. As well as the epidemiological design advantages in terms of adjusting for confounders, these longitudinal studies provide risk estimates for increases in incidence rather than in prevalence. This simplifies the flow-through to economic valuation.
4.4.2 Mortality

Mortality effects of chronic exposure to air pollution have frequently been studied using cross-sectional designs; and especially in population based (or ecologic) cross-sectional studies which correlate city mortality rates with ambient air pollution concentrations. These studies generally reported positive associations for particles. Their major limitation lies in their ecologic/design; that is, aggregated measures are used for groups of individuals. This approach does not permit control for individual differences in confounders, such as cigarette smoking. The strength of the relationship found was often sensitive to model specification and to socio-economic, demographic, and other risk factors, as well as to the choice of study area. These cross-sectional studies were comprehensively reviewed by Lipfert (1994) who concluded that although suffering from methodological flaws they showed consistent associations between particles and mortality at different times and places, and consistent in a positive direction.

Cross-sectional differences in mortality and air pollution have also been studied in prospective cohort studies (Dockery et al 1993, Pope et al 1995c). In these studies the characteristics of the subjects, including their smoking habits were collected on an individual basis.

Dockery et al (1993) used data from the Harvard Six-Cities Study involving six cities in the USA with differing levels of air pollution; Steubenville, Ohio being the most polluted city and Topeka, Kansas, the least polluted. The study cohort consisted of approximately 8,000 people followed up for 14 to 16 years between 1974 and 1991. The prospective cohort design allowed for control of individual differences in age, sex, cigarette smoking, education level, occupation and BMI. Total suspended particles (TSP), PM$_{10}$, PM$_{2.5}$, SO$_4$, H$^+$, SO$_2$, NO$_2$ and ozone levels were monitored throughout follow-up as part of the Six-Cities study. Using Cox regression methods and adjusting for individual differences showed that adjusted mortality rates for Steubenville were 26% higher (95% CI, 8%, 47%) than those for Topeka. For various causes of death, the six adjusted city-specific death rates were then compared with city-specific indices of pollution. Contrasts between six cities only might be expected to give little discriminatory power in establishing which pollutants might be involved, or the shape of the exposure-response relationship. However, the adjusted death rates were significantly associated with particles (PM$_{10}$, PM$_{2.5}$) and sulphates (SO$_4$), but not with ozone. The particle association was consistent, approximately linear and showed no threshold effect.

In a larger prospective study of 500,000 individuals in 151 cities, similar results were observed over a follow up period of 8 years (Pope et al 1995c). An association was found between mortality and air pollution after adjusting for age, sex, race, smoking, exposure to passive smoking, occupational exposure, education, BMI, and alcohol use. The adjusted mortality rate was 15%-17% higher in the most polluted areas compared to the least polluted in terms of sulphates and fine particles (PM$_{2.5}$). The strongest associations were observed for cardiopulmonary disease and lung cancer, with insignificant associations with death due to other causes.
4.4.3 The Seventh-Day Adventists studies

Populations of non-smokers provide an opportunity to study health impacts of air pollution, where the confounding of the relationship by smoking status is removed or at least diluted. One such study by Euler et al (1987) was based on an adult population of Seventh-Day Adventists who have low consumption of tobacco and alcohol. Euler et al (1987) considered 7,445 adults (aged 25+) of this population who had been exposed to air pollution in California for at least 11 years. Cumulative exposures for SO2 and TSP were estimated using resident codes and interpolated dosages from state air monitoring stations. After allowing for occupational exposure, passive smoking, ever smoking, age, sex, race and education in a logistic analysis, statistically significant associations of COPD with cumulative SO2 and cumulative TSP were found. The association with COPD with TSP was stronger than that for SO2.

Abbey et al (1995) gave an overview of a number of studies of seventh day adventists, including those by Euler et al (1987) and by Abbey et al (1993), and assessed which pollutant was most strongly associated with respiratory disease. All the studies reviewed by Abbey et al (1995) were based on follow-up of 6340 non-smoking Seventh-Day Adventists and who had lived within 5 miles of their present residence for at least 10 years. Indoor sources of NO2, tobacco smoke at home and at work, as well as occupational exposure to dust and fumes were taken into account in the statistical analyses. Pollutants considered were TSP, PM10, PM2.5, SO2, SO4, NO2 and ozone. There were no statistically significant associations between disease outcomes and NO2 or SO2. There were also no significant associations between the pollutants and all cause mortality or all cancers in males, although TSP and PM2.5 were associated with all cancers in females. The effects of TSP, PM10 and PM2.5 could not be truly separated because of the way in which they were measured and also because of their high correlations. Statistically significant associations were found between one or more of the particle measures and airway obstructive disease (AOD), chronic bronchitis and asthma with TSP showing the strongest association with AOD and chronic bronchitis. SO4 was significantly associated with development of asthma, an association which was stronger than that for TSP or ozone. Ozone was significantly associated with the development of asthma in males only, increasing severity of asthma and with respiratory cancer. Multi-pollutant analyses suggested that none of the associations found were due to correlations with other pollutants.

4.4.4 Other chronic morbidity

In considering other studies of the chronic effects or cumulative effects of air pollution on respiratory symptoms and lung function, we pay particular attention to those studies based on individuals, where adjustment for confounding is at the individual level also.

(a) Lung Function: Lipfert (1994) gives a detailed review of lung function studies. European studies since 1980 discussed by Lipfert include a major EC study of children aged 6-11 (Florey et al, 1983), where the authors considered that confounding between countries had not been overcome satisfactorily; the PAARC study in France (PAARC 1982b), where re-analyses by Lipfert indicated a relationship with one measure of SO2; and a smaller study of
children in Turin (Spinaci et al, 1985; Arossa et al, 1987) which examined changes over a two-year period.

Central to any evaluation of the chronic effects of air pollution on lung function are two studies using data obtained from the US National Health and Nutrition Examination Survey (NHANES: I and II) (Chestnut et al, 1991, Schwartz 1989). Chestnut et al (1991) examined TSP only in studies of adults using data from NHANES I, 1971-75. They found statistically significant relationships between TSP and lung function (FVC, FEV1) in adults after adjusting for age, sex, height, race, obesity, smoking weather and socio-economic variables. The analysis suggested a threshold for the effects of TSP at approximately 60 µg/m³, below which a relationship with lung function was no longer found.

Schwartz (1989) used data from NHANES II, 1976-80, to consider the effect of air pollution on lung function in children and young people aged 6 to 24 years. In an analysis described by Lipfert (1994) as ‘in many ways a paradigm for deriving estimates of the effects of air pollution on lung function [in cross-sectional studies]’, Schwartz found significant negative relationships between FEV1, and FVC and annual TSP, ozone and nitrogen dioxide, allowing for age, height, race, sex, body mass index, cigarette smoking and respiratory conditions. These relationships were similar for most concentrations of NO₂, but only affected at high levels of ozone (>0.04 ppm) and TSP (>90 µg/m³), and hence suggest a threshold for effects of ozone and TSP. Lung function was also studied by Vedal et al (1991).

The lung function endpoints are not of direct relevance to the ExternE project at this time, because economic valuation figures are not available. However, they supply context to the wider assessments.

(b) Respiratory Symptoms in Adults: Schwartz (1993b) looked at the association between chronic respiratory disease and particulate air pollution, using data from NHANES I (1971-75). Analysis included 3874 subjects aged 30 years and over, from 53 urban areas. Using logistic regression methods, and adjusting for age, race, sex and smoking, Schwartz found that annual average TSP concentrations were significantly associated with chronic bronchitis and with respiratory illness. The odds ratios increased for never smokers alone suggesting that the association was not due to lack of control for smoking. There was no significant association with asthma.

(c) Children: Dockery et al (1989) used data collected in the school year 1980-81 to (re)-examine the relationships between respiratory illness and particles in a cohort of 5422 white school-children aged 10-12 years, surveyed annually as part of the Harvard Six-Cities Study. These follow-up examinations included a respiratory symptom questionnaire completed by a parent and a spirometric examination carried out at home.

Annual average concentrations of SO₂, NO₂, O₃, PM₁₀, TSP and PM₂.₅ for the 12 months preceding the examination were analysed in relation to each of five respiratory outcomes: doctor’s diagnosis of bronchitis in the last year, chronic cough present for at least 3 months in the last year, chest illness which restricted activity for at least 3 days, persistent wheeze and doctor diagnosis of asthma. A two step analysis was used to account for city to city variation in health outcomes. Initial logistic regressions modelled the probability of illness in each of
the six cities separately, controlling for sex, age, maternal smoking and the presence of a gas stove in the home. In the next step, the estimated logits were regressed against city-specific air pollution measurements using weights inversely proportional to the sum of the between city variance and the within-city variance of the adjusted logits.

The results of the regression were expressed as odds ratios of illness in the most polluted cities relative to the least polluted cities. Asthma was significantly related to \( \text{O}_3 \). \( \text{PM}_{10} \) was significantly related both to bronchitis and to chronic cough. The associations with air pollution stronger in those with a history of wheeze or asthma.

### 4.4.5 Interim conclusions

It is useful to recap on the main methodological limitations relevant to quantitative risk estimates from the studies of chronic effects. A major limitation lies in the lack of good information on the true exposures of individuals in biologically relevant time periods; and the consequent use of surrogates based on long-term (or recent annual) average concentrations for a wide area (e.g., a city). The difficulty of adjusting appropriately for confounding factors, and/or the lack of longitudinal studies which take account in the design of confounders between individuals, is a second major limitation. Migration of individuals from one area to another adds to misclassification error. It is also difficult to disentangle chronic effects from the accumulation of acute effects.

There is nevertheless substantial qualitative evidence and, increasingly, more reliable quantitative evidence also that longer-term exposure to ambient air pollution may increase the risks of chronic disease and premature death. Given this evidence, and the importance of the effects both in terms of loss, or deterioration of quality of life and its associated costs, we consider it worthwhile to attempt to quantify both chronic mortality and chronic morbidity effects of the various fuel cycles; with the understanding however that the strength of evidence is substantially less than for acute effects, and needs to be kept under review.

### 4.5 Summary of Model Assumptions

Quantifying the public health effects of ambient air pollution from power generation can be viewed as the development and application of a risk model which had two principal aspects:

(a) A set of model assumptions or strategic judgements which provide a broad framework for the implementation: and

(b) Within that framework, a detailed set exposure-response functions linking particular pollutants with specific health endpoints; together with other details of the methods of implementation.

It is convenient at this point to summarise the principal model assumptions that arise from the literature reviews; and then, in subsequent sections, describe the specific E-R functions used.
4.5.1 Particles: acute effects

There are now numerous well-conducted studies linking particulate air pollution with a wide range of acute health effects with no convincing evidence of a threshold level. There is some evidence, though not compelling, that the health effects per unit incremental exposure are higher when background pollution levels are lower. Because this is not well-established, and for ease of implementation, effects are estimated in this project as independent of background; and in particular, without threshold.

There is a growing tendency to treat the associations as causal, though the mechanism of action is unknown. It is not known for sure what size range or what components of respirable particulate air pollution are responsible. It seems however that particles emitted from combustion sources, or formed subsequently (e.g. sulphates, nitrates) are more dangerous to health than wind-blown natural (crustal) particles.

Within the present project, incremental particulate air pollution may arise due to the direct emissions of particles during power generation; and/or due to the subsequent formation of sulphates (from SO\textsubscript{2}) and nitrates (from NO\textsubscript{X}). We have followed a widespread current convention in using PM\textsubscript{10}, i.e. particles of less that 10 µg/m\textsuperscript{3} aerodynamic diameter, as the most relevant index of ambient particulate concentrations; and in applying relationships for PM\textsubscript{10} to sulphates and to nitrates as well as to particles directly emitted. A case can however be made for relationships based on sulphates or on fine particles (PM\textsubscript{2.5}). The estimated effects of sulphates would then be higher, perhaps by a factor of, say, between two and four. The situation should be kept under review, as evidence develops.

4.5.2 Oxides of nitrogen

Some studies link NO\textsubscript{X} or NO\textsubscript{2} with acute effects. These studies have not been used, because the apparent NO\textsubscript{X} effect is arguably in reality not an effect of NO\textsubscript{X} as such. Rather, NO\textsubscript{X} may be a surrogate or marker for a mixture of pollutants not otherwise well measured in these studies, including particles from combustion sources, especially traffic. Some role for NO\textsubscript{X}/NO\textsubscript{2} is not discounted, but the direct effect if any is likely to be small and not quantifiable reliably on the basis of available studies. NO\textsubscript{X} is however implicated indirectly via nitrates and ozone.

4.5.3 Sulphur dioxide

Similarly, with SO\textsubscript{2}, we have assessed indirect effects via sulphates, but have not ascribed acute health effects to SO\textsubscript{2} as such. The range and diversity of positive studies linking SO\textsubscript{2} with acute health effects is quite substantially greater than for NO\textsubscript{X}, and human experimental studies are more suggestive of a real link, especially for asthmatics. The position nevertheless is ambiguous. Arguably, SO\textsubscript{2} in itself is not the cause of the observed epidemiological associations. In several studies, apparent SO\textsubscript{2} effects disappear when particles are measured appropriately. Hence our decision. Again, further work and ongoing review are needed, especially in the light of new results from studies in Europe (the APHEA project) which are just now coming available.
4.5.4 Ozone: acute effects

There are positive studies linking ozone with a wide range of acute health effect endpoints. Most of these are summertime studies, in high-ozone areas. It seems that a real relationship exists in these circumstances, and exposure-response relationships are proposed accordingly; though the evidence is much stronger, and from a greater diversity of locations, for some endpoints (e.g., respiratory hospital admissions) than for others (e.g., mortality). Daily 1-hr maximum ozone concentration is used, though many recent studies use longer time-periods which possibly are more biologically relevant. Most of these studies do not suggest a threshold for ozone effects. It is nevertheless unclear whether these results generalise to other locations and seasons, where direct investigation often fails to show an ozone effect. We treat the results as generalisable, without threshold, on the assumption that ozone is the active agent. However, this uncertainty about generalising, together with difficulties of ozone modelling, have led to implementation only where background concentrations are higher than 60 ppb.

4.5.5 Particles: chronic effects

Recent studies, adjusting for confounders at the individual level, have shown associations between chronic health effects and ambient particulate levels, though the adequacy of adjustment for confounders remains a matter of debate. Quantification of the chronic effects is complicated, and compromised, by use of recent pollution levels only, rather than the possibly higher historical levels which may be biologically relevant to chronic mortality and morbidity. (Use of recent levels only will over-estimate the risk per unit exposure to an unknown extent.)

Evidence of a chronic effect on mortality is given by Dockery et al (1993) and Pope et al (1995). These two cohort studies found clear relationships both with fine particles (PM$_{2.5}$) and with sulphates. Risk estimates from Pope et al (1995), ‘translating’ from PM$_{2.5}$ to PM$_{10}$, are used as the best available estimate of chronic mortality. They show that chronic mortality effects may have a major impact on the overall evaluation. However, there is general agreement at present that quantified estimates of chronic mortality may be unreliable. Consequently, these estimates are not incorporated into summary tables.

Some aspects of chronic morbidity are also evaluated. These have been included, on the assumption that morbidity is less affected by the ‘historical-to-recent’ exposure problem. This is certainly true for childhood chronic disease. It is arguable whether the childhood conditions estimated here should be called ‘chronic’ or ‘acute’.

4.5.6 Other pollutants: chronic effects

The situation regarding chronic effects of ozone is unclear. There may be some effects, for example on asthma severity. However, relationships are not proposed. Nor are chronic effects estimated for SO$_2$ or NO$_x$. 
4.5.7 Summary of exposure-response relationships proposed

Effects quantified now include mortality, hospital admissions, (hospital) emergency room visits (ERVs), restricted activity days (RADs), asthma attacks and (mostly respiratory) symptoms. Lung function effects are not described because there is currently no good means of economic valuation.

The relationships proposed are described in the following Sections. For each E-R relationship we report a ‘mid’ or best estimate of effects, together with a ‘high’ and ‘low’ estimate based on estimates that are respectively 1 SE higher or lower than the ‘mid’ estimate. As discussed later, this quantifies only one aspect of the uncertainties involved in the estimation process as a whole.

4.6 Exposure-Response (E-R) Relationships: Acute Health Effects

4.6.1 Mortality

(a) Choice of papers

There are now well-conducted time-series epidemiological studies in about 15 locations, mostly but by no means exclusively in the USA, giving risk estimates linking daily deaths with daily particulate air pollution, having adjusted for season, weather and other confounding factors (Section 4.2, above; Dockery and Pope, 1994; Pope et al, 1995a; 1995b). Converting the index of particles to PM$_{10}$, these studies generally report risks in the range of 0.5% to 1.5% increase in mortality per 10 µg/m$^3$ increase in daily PM$_{10}$.

We have based our risk estimates on one study from Birmingham, Alabama, USA (Schwartz, 1993a). This is by no means the most comprehensive of the studies available; or the most appropriate in terms of circumstances studied. For example, most mortality studies in the USA had been carried out in the North or West, whereas Birmingham, Al. provides a chance to investigate associations with low concentrations of particles in Southern USA. As in the most of eastern USA (but unlike in Europe), particle concentrations are highest in the warm months and mortality is highest in the cold months, so confounding by season would be unlikely.

The risk estimates from Birmingham (Al.) are however near the centre of those from the range of studies available. Furthermore, the SE of the risk estimate is quite large compared with other studies, reflecting that this particular study was among the less powerful in identifying an association. For our purposes however this has an advantage compared with using, say, Schwartz and Dockery (1992b), where the risk estimate is similar but its SE is much smaller. Using the Birmingham study, with its greater uncertainty, is a device to take some account also of the uncertainty in transferring relationships from the USA to Europe. Finally, the Birmingham study used PM$_{10}$ as its index of particles; and so there is no need to ‘translate’, say from TSP to PM$_{10}$. 
Public Health Effects

There are few studies reporting relationships between daily deaths and ozone concentrations. Kinney and Ozkaynak (1991) is a detailed report; but is based on total oxidants, not ozone. Kinney et al (1994) gives an update, using ozone; and so, though available in Abstract only, this is our choice of paper, as the methods also appear to be very similar to those described in detail by Kinney and Ozkaynak.

(b) Particles (PM₁₀) and acute mortality: Schwartz (1993a)

Schwartz (1993a) considered the association between daily mortality 1985-1988 particles (PM₁₀) in Birmingham, Alabama, a metropolitan area with a population of 884,000. Daily 24-hour average levels of PM₁₀ from the one or two available monitors within the city were calculated. Other pollutants were not included.

A thorough approach to specification of the model was considered in the statistical methods used. Poisson regression was used to model the daily counts of deaths. The method of generalised estimating equations of Liang and Zeger (1986) was incorporated to account for overdispersion and serial correlation. Long term patterns were filtered out of the daily deaths by adding 24 sine and 24 cosine terms to the regression model, covering all cycles with periods from 2 years to 1 month. Also annual dummy variables and linear and quadratic time trends were added to capture any temporal trends. Finally, dummy variables for day of the week were used to capture weekly patterns.

Initial analyses of the weather effect on mortality considered variables for 24 hour mean temperature, dew point temperature, dummy variables for humid days, hot days, cold days and hot and humid days. Only after the baseline model with weather terms was established was the air pollution effect in predicting daily mortality examined.

A number of strategies were used to assess whether the association between particles and daily mortality was sensitive to the specified model. Firstly, the use of the Poisson model was compared to ordinary least squares regression, and also to robust regression which gives unbiased estimates of the regression coefficients standard error. Secondly, the adequacy of the weather model was examined using generalised additive models, which help to detect non-linearities in these relationships. This involves fitting smoothed curves which consist of weighted moving averages, with the weights declining with distance from the centre of the neighbourhood. Thirdly, an alternative filtering method was assessed which used non-parametric smoothing. This method more readily captures non-sinusoidal relationships and is more parsimonious compared to the trigonometric filter.

Having controlled for weather, time trends, day of the week, year of study and removed long term trends by filtering, a significant association was found between particles (PM₁₀) and daily mortality (RR = 1.11, 95% CI = 1.02, 1.20) for an increase of 100 µg/m³ PM₁₀.

Essentially identical results were obtained when least squares regression was used, when robust regression was used and under the alternative filtering method. There was no evidence of a threshold effect down to the lowest observed particle levels. The association was also unchanged when days of PM₁₀ level above the statutory standard were excluded from the analysis. Thus, the results were robust to the statistical model used.
The RR of 1.11 (95% CI = 1.02, 1.20) per 100 µg/m$^3$ PM$_{10}$ implies a Poisson regression coefficient of 0.001044 (se. 0.0004062) per µg/m$^3$ PM$_{10}$; and these in turn imply estimates of percentage change in mortality of

<table>
<thead>
<tr>
<th></th>
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<th>per µg/m$^3$ PM$_{10}$</th>
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</thead>
<tbody>
<tr>
<td>Low:</td>
<td>0.0638 %</td>
<td></td>
</tr>
<tr>
<td>Mid:</td>
<td>0.1044 %</td>
<td></td>
</tr>
<tr>
<td>High:</td>
<td>0.1450 %</td>
<td></td>
</tr>
</tbody>
</table>

(c) Ozone and acute mortality: Kinney et al (1994)

Both Kinney and Ozkaynak (1991) and Kinney et al (1994) have been described in detail in 4.3.2, above; the detail on Kinney et al (1994) being of course limited by the fact that only an Abstract was available. The increase in deaths was estimated as 0.024 (se. 0.008) deaths per day per ppb O$_3$. The size of the at-risk population is not given. However, Kinney et al (1994) report that, in percentage terms, the increase is approximately 1.1% for 72 ppb ozone, the mean level during the study period. This is approximately equivalent to a 0.015% increase (se. 0.005%) per ppb O$_3$; giving estimates of percentage change in mortality of

<table>
<thead>
<tr>
<th></th>
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<th>per ppb ozone</th>
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<tbody>
<tr>
<td>Low:</td>
<td>0.010 %</td>
<td></td>
</tr>
<tr>
<td>Mid:</td>
<td>0.015 %</td>
<td></td>
</tr>
<tr>
<td>High:</td>
<td>0.020 %</td>
<td></td>
</tr>
</tbody>
</table>

4.6.2 Hospital admissions

(a) Choice of endpoints

Several studies have shown that acute respiratory hospital admissions are related both to particles and to ozone. Given the findings on acute mortality, it might be suspected that acute admissions for cardiovascular causes are also related to ambient pollution levels, especially for particles (and possibly not for ozone). There has however been little or no investigation of this issue. We understand that some US studies are currently in progress which link daily acute cardiovascular admissions with daily particulate air pollution. However, results were not available for inclusion in the present exercise. The situation should be kept under review.

Air pollution affects the numbers both of daily deaths and daily hospital admissions. It seems likely therefore that it also affects the severity of outcome of some people who would in any case have been admitted to hospital but who survive; i.e., of people who are not included in the outcomes considered here. That severity might be marked, for example, by length of stay in hospital. We know of no studies which have examined this issue and so severity of condition post-hospitalisation is not quantified (except insofar as death may result soon afterwards).
It would have been possible to treat ‘acute respiratory admissions’ as an outcome to be evaluated. However, several studies have now been reported which disaggregate this outcome into component subgroups of subsidiary causes. The evidence suggests that infections (notably pneumonia) and chronic obstructive pulmonary disease (COPD) are positively associated with both particles and ozone. The evidence regarding acute attacks of asthma is less clear, partly because numbers to be studied are usually lower than for the other subgroups. It appears however that the evidence linking asthma admissions with ozone is stronger than with particles; and we have quantified impacts accordingly.

(b) Strategy; and choice of source papers

The effects of particles (PM$_{10}$) and ozone appear to be additive within a Poisson regression framework; i.e., when expressed as percentage increases over background levels of hospital admissions. There is however some safeguard in estimating effects of both pollutants from the same study and the same regression model, so that the effect of each is adjusted for inclusion of the other. There is also a benefit in using, if practicable, a study where particles were measured as PM$_{10}$ (to avoid the need for conversion from some other index of particulate pollution). Several papers qualify. Because its methodology is well-described and because it is one of a series of key hospital admissions papers, we have chosen Schwartz (1994d) for admissions for pneumonia and COPD in relation both to PM$_{10}$ and to ozone; and Thurston et al (1994b) for asthma admissions and ozone.

An alternative to choosing individual studies is to carry out a formal or semi-formal meta-analysis of results from several studies. This might be considered in future revisions, though we think that results would be similar.

Schwartz (1994d) studied hospital admissions for the elderly only. However, the age-specific results of Burnett et al (1994) suggest that the percentage increase in hospital admissions is similar across a wide range of age-groups and arguably for all ages. We have therefore applied the percentage increases from Schwartz (1994d) to all-ages background rates of admissions based on Burnett et al (1994).

Thurston et al (1994b) report their regression results in terms of estimated additional impacts per day per unit exposure; i.e., directly in the units which we would wish to use. Their study however uses data from six-week periods during July/August only; and background admission rates for asthma in this short period might well be untypical of the year as a whole. We therefore re-expressed the Thurston et al findings in terms of percentage change per unit exposure; and linked these percentage increments with the background admission rates from Burnett et al, as before.

(c) Hospital admissions for pneumonia and for COPD: Schwartz (1994d).

The relationships between ambient pollution (from PM$_{10}$ and from ozone, but not from SO$_2$ or NO$_x$) and hospital admissions for various respiratory causes in the elderly in Detroit, Michigan were analysed by Schwartz (1994d). Data for the years 1986 through 1989 were
available for the population, of whom 517,000 were aged 65 years and over in 1990. Daily counts of admissions for pneumonia (ICD9, 480-486) and COPD (ICD9, 490-496) were extracted, along with asthma (ICD9 493), from the US Medicare insurance records of all hospitals in the Detroit statistical area. Particles, measured as \( \text{PM}_{10} \), are 24-hr averages, in units of \( \mu g/m^3 \). Ozone was measured both as 24-hr average and as a 1-hr daily peak and expressed as ppb. The analyses were based on these indices averaged over all monitoring stations in the Detroit statistical area (2-7 stations for \( \text{PM}_{10} \), 8 or 9 stations for \( O_3 \)), with exclusions for missing data. Schwartz (Discussion, p653) reports that on average, peak 1-hr ozone was 2.8 times as high as the 24-hr. daily average. The correlation of 0.35 between \( \text{PM}_{10} \) and ozone was not particularly high.

Analyses of the daily admission counts were carried out using Poisson regression with allowance for over-dispersion. Before considering the effects of air pollution, a baseline model was developed as follows. Dummy (indicator) variables for each of the 48 months of follow-up were used to take account of seasonal variations in respiratory admissions. In addition, linear and quadratic time trends were included. Dummy variables (eight each for categories of daily mean temperature and dew point temperature) were also used to take account of possible confounding by weather in a way that avoids specifying the shape of this relationship. (Non-parametric smoothing methods were also used and gave similar estimates of relationships with air pollutants.)

Residual autocorrelation was also considered and, where appropriate, adjusted for using generalised estimating equations (Liang and Zeger 1986). However, no significant autocorrelation was found.

Within this framework, statistically significant relationships were found linking daily admissions for COPD and pneumonia with same-day \( \text{PM}_{10} \) and previous-day ozone. Controlling for one pollutant did not importantly affect the magnitude of the association with the other; nor did the exclusion of high-pollution days (\( \text{PM}_{10} > 150 \mu g/m^3 \) or \( O_3 > 120 \text{ ppb} \)). No corresponding significant relationships with asthma were found and asthma results are not reported in detail.  

The basic model of Table 4 of Schwartz (1994d) gives regression coefficients for pneumonia on \( \text{PM}_{10} \) (per \( \mu g/m^3 \)) and on ozone (per ppb: 24-hr average values) of 0.00115 (SE 0.00039) and of 0.00521 (SE 0.0013) respectively. Corresponding coefficients for COPD were 0.00202 (SE 0.00059) for \( \text{PM}_{10} \) and 0.00549 (SE 0.00205) for ozone. As these results were obtained from Poisson modelling, they can be interpreted as the percentage increase giving Low, Mid and High values as shown below.

<table>
<thead>
<tr>
<th></th>
<th>Pneumonia</th>
<th>COPD</th>
</tr>
</thead>
<tbody>
<tr>
<td>Percentage increase in daily admissions per ( \text{PM}_{10} ) (( \mu g/m^3 ))</td>
<td>Low 0.076%</td>
<td>Low 0.143%</td>
</tr>
<tr>
<td></td>
<td>Mid 0.115%</td>
<td>Mid 0.202%</td>
</tr>
<tr>
<td></td>
<td>High 0.154%</td>
<td>High 0.261%</td>
</tr>
</tbody>
</table>
These are the percentage increases used in the implementations of the various fuel cycles. However, use of the ozone values within the later implementation requires that they be expressed in terms of effects per ppb (1-hr peak) rather than per ppb (24-hr average) as given above. This involves scaling down by a factor of 2.8 to give:

<table>
<thead>
<tr>
<th></th>
<th>Pneumonia</th>
<th>COPD</th>
</tr>
</thead>
<tbody>
<tr>
<td>% increase in daily admissions per ozone (ppb):</td>
<td>Low 0.140%</td>
<td>Low 0.123%</td>
</tr>
<tr>
<td>24-hr averages</td>
<td>Mid 0.186%</td>
<td>Mid 0.196%</td>
</tr>
<tr>
<td>1-hr maxima</td>
<td>High 0.232%</td>
<td>High 0.269%</td>
</tr>
</tbody>
</table>

(Note/Erratum: In error, the unadjusted figures were used in the analysis presented in the ExternE coal report (European Commission, 1995), giving estimated hospital admissions 2.8 times too high, per ppb incremental ozone expressed as a 1-hr peak.)

(d) Hospital admissions for asthma: Thurston et al, 1994b.

Thurston et al (1994b) considered the acute effects of ozone and of several other pollutants, notably acid aerosols, on hospital admissions for asthma (ICD 493) and for other respiratory conditions in Toronto, Ontario. Daily admissions from 22 acute care hospitals (population 2.4 million) were obtained for a six week period in summertime for 1986-1988. Hourly observations of \( O_3 \) were available from one site only. For consistency with other studies, 1-hr daily maximum was used as the \( O_3 \) index of choice in the statistical analysis.

Following extensive data descriptions, statistical regression analyses used hospital data that had been adjusted to remove seasonal effects (by regressing on sine and cosine curves of annual cycles) and day-of-the-week effects (by regressing the detrended residuals on appropriate dummy variables). The resulting residuals were approximately Normally distributed and so ordinary least squares, rather than Poisson regression, was used subsequently. The temperature lag having highest association with the relevant hospital admission series was then included before the effects of various pollutants were examined, individually and in combination.

Table 4 of Thurston et al (1994b) gives the regression coefficient for ozone on daily admissions for asthma adjusting for temperature and particles (\( PM_{10} \)) as 0.0290 (SE= 0.0146) admissions per ppb increase in ozone. The mean daily admission rate for asthma was 8.93 per day and so the regression coefficient represents, in terms of percentage change, an increase of 0.029/8.93 = 0.325% (low = 0.161%, high = 0.488%) in daily admissions for asthma.
Burnett *et al* (1994) studied daily respiratory hospital admissions in 168 acute care hospitals in Ontario, Canada, 1983 through 1988. The study region included large (Toronto-Hamilton) and medium-sized (Ottawa, London, Windsor) cities, small communities and large rural areas. The at-risk population was about 8.7 million people. Burnett *et al* found that the rate of daily admissions for pneumonia/infection was 38.8 per day and for COPD was 26.8 per day. (There were minor differences in how Schwartz (1994d) and Burnett *et al* (1994) defined respiratory subgroups in terms of ICD codes. For example, Burnett *et al* (1994) included ICD 466 (acute bronchitis) in the infection/pneumonia category. We have used respiratory infection and pneumonia interchangeably in the subsequent text and tables.)

The average daily admissions for asthma among all 168 hospitals was 41.9 which is equivalent to $41.9/8.7 = 4.81$ per million per day = 175.6 per 100,00 per year.

### (f) Estimated annual impacts per pollution increment

Applying the percentage increases summarised above from Schwartz (1994d) to the daily admission rates for *all* ages obtained from Burnett *et al* (1994), and annualising, gives estimates of additional impacts per increment of exposure as shown in the following Tables. For example, the estimated increase in pneumonia admissions is calculated as $38.8/87 \times 0.00115 \times 365 = 0.187$ per 100,000 per year per PM$_{10}$ ($\mu$g/m$^3$).

| Increase in admissions for COPD per 100,000 population per year | Low 0.161 | Change in Annual PM$_{10}$ ($\mu$g/m$^3$) |
| Increase in admissions for pneumonia per 100,000 population per year | Low 0.124 | Change in Annual PM$_{10}$ ($\mu$g/m$^3$) |

| Increase in admissions for COPD per 100,000 population per year | Low 0.387 | Change in Annual Ozone (ppb) |
| Increase in admissions for pneumonia per 100,000 population per year | Low 0.636 | Change in Annual Ozone (ppb) |

These are the values used in the ExternE implementations. However, the ozone values should have been adjusted downwards by a factor of 2.8 in ‘translating’ from effects per ppb (24-hr average) to effects per ppb (1-hr daily maximum). The corrected values are:
Public Health Effects

| Increase in admissions for COPD per 100,000 population per year | Low 0.138 | Change in Annual * Ozone (ppb) |
| Increase in admissions for pneumonia per 100,000 population per year | Mid 0.220 | 1hr daily max |
| | High 0.303 | |

Applying the asthma results from Thurston et al (1994b) to the background admission rates from Burnett et al (1994) implies the following results:

| Increase in hospital admissions for asthma per 100,000 population per year | Low 0.283 | Change in Annual * Ozone (ppb) |
| | Mid 0.571 | 1hr daily max |
| | High 0.858 | |

These asthma estimates are as used in ExternE and are in the correct units for that implementation.

4.6.3 Emergency room visits

(a) Choice of endpoints and of source papers

As with hospitals admissions, for emergency room visits (ERVs) we looked for, and used, exposure-response relationships for specific respiratory endpoints rather than for ‘respiratory causes’ as a whole. In general, there are far fewer good studies of ERVs than there are of hospital admissions. Indeed, there were no suitable papers giving exposure-response relationships simultaneously for particles and for ozone. Relationships were found from well-conducted studies quantifying the effect of particles on ERVs both for chronic obstructive pulmonary disease (COPD) and for asthma; and of ozone on asthma only. These specific respiratory ERV endpoints are broadly consistent with those for other acute respiratory effects. It is unclear whether in reality there is also an acute effect of ozone on ERVs for COPD; the asthma/ozone relationship has been studied much more intensively.

The papers selected to quantify the effects of PM$_{10}$ and of ozone on ERVs are as follows:

i. Sunyer et al (1993) link particles with ERVs for COPD. This study uses black smoke rather than PM$_{10}$ as the index of particulate air pollution; hence, ‘translation’ to PM$_{10}$ is necessary. It updates the earlier paper by Sunyer et al (1991) and provides results in a way that can be expressed as number of additional ERVs per year per µg/m$^3$ PM$_{10}$ per 100,000 people at risk.
ii. Schwartz et al (1993) links PM$_{10}$ with ERVs for asthma. This provides information on percentage change directly in terms of PM$_{10}$ but does not report population data. As for hospital admissions, background rates were taken from another study; in this instance, Bates et al (1990).

iii. Schwartz et al (1991) is a study in Germany of children’s visits to hospitals and/or to paediatricians; and so incorporates a very specific endpoint for one susceptible subgroup of the population. The endpoint appears to be a mixture of ERVs and visits to a doctor; but classification as an ERV seems the more appropriate. Baseline rates can be obtained from within the paper itself.

iv. Cody et al (1992) link ozone and ERVs for asthma. Again, the information relates to percentage change only; and background rates are estimated from Bates et al (1990).

(b) PM$_{10}$ and ERVs for COPD: Sunyer et al, 1993.

Sunyer et al (1993) studied the daily number of ‘emergency room admissions’ for chronic obstructive pulmonary disease (COPD) among residents aged 14 years or more in Barcelona in the period 1985-1989; in relation to daily levels (24-hr average) of black smoke (reflectometry) and SO$_2$ (conductivity method): average of 15 manual samplers throughout the city. The study was based on data from the four largest hospitals in Barcelona. An emergency room admission was defined as in Sunyer et al (1991); i.e., as ‘a visit during which any diagnosis related to COPD was recorded’; a definition which corresponds more closely to ERVs than to hospital admissions as such. Most (96%) of the ERVs for COPD studied referred to people aged over 45 years.

Analyses were carried out separately for winter and for summer, the daily average number of ERVs for COPD being 15.8 in winter and 8.3 in summer. Controlling for season, meteorology and autocorrelation, statistically significant associations were found linking ERVs for COPD in winter with both sulphur dioxide and black smoke, included separately and jointly. The size and statistical significance of the effects was markedly lower when both pollutants were included. A weaker and not statistically significant association between black smoke and ERVs for COPD in summer disappeared when SO$_2$ was also included in the model; the estimated SO$_2$ effect being much closer to its wintertime value.

Clearly, this study shows an effect of air pollution on ERVs for COPD which should be taken into account. However, its results sit uneasily with the strategic decision to quantify relationships based on particles and not on SO$_2$. Because of that strategic decision, we decided to use relationships with black smoke unadjusted for SO$_2$ as the best means of expressing the air pollution effect; and on the general grounds that the apparent SO$_2$ effects might in reality reflect aspects of particles not accounted for by black smoke (though other analyses, not reported in detail, found that adjustment for sulphates, for NO$_2$ or for ozone did not account for the SO$_2$ effects). Furthermore, because the relationship with black smoke does not express well the summertime pollution effect in this study, we have assumed that the somewhat higher percentage effect of winter applies in summertime also.
The winter-time increase in the number of ERVs for COPD was estimated as 0.90 per day for an increase of 25 µg/m$^3$ in black smoke, in models not including SO$_2$. The associated t-value of the regression coefficient was 5.0, implying a SE of 1.8. As the mean number of visits for COPD in winter was 15.8, these coefficients represent an increase of 5.70% (SE = 1.14%) giving low and high values of 4.56% and 6.84% respectively for 25 µg/m$^3$ increase in black smoke.

In ‘translating’ to PM$_{10}$, we have followed Dockery and Pope (1994) in assuming an equivalence between black smoke and PM$_{10}$. (In reality of course the relationship differs according to the specifics of the pollution mixture and this conversion factor might usefully be re-examined.)

The daily wintertime and summertime rates for ERVs for COPD imply an annual number of \((15.8 + 8.3/2) \times 365 = 4398\) visits per year. The population of Barcelona in 1985-86 was about 1.7 million people (Sunyer et al, 1991). The four hospitals studied covered ‘more than 80 percent of the respiratory emergencies in the city’ (Sunyer et al, 1993), implying an effective population at risk of about 1.4 million people at all ages. This implies a mid-value estimate of an annual increase of \((0.057 \times 4398)/(25 \times 14) = 0.72\) per 100,000 per µg/m$^3$ PM$_{10}$, with ‘low’ and ‘high’ values differing by minus or plus 20%, respectively. Thus,

\[
\text{Increase in ERV per year} = \text{Mid 0.72 Change in per COPD per 100,000} = \text{Low 0.58 Annual PM}_{10} \text{High 0.86 (µg/m}^3\text{)}
\]

(c) PM$_{10}$ and ERVs for asthma: Schwartz et al (1993) and Bates et al (1990).

Schwartz et al (1993) considered the relationship between exposure to particles (PM$_{10}$) and ERV for asthma. The data were obtained from eight hospitals in the Seattle area, USA. Asthma was defined as ICD-9 codes 493, 493.01, 493.10, 493.90 and 493.91. There was a total of 2955 visits for asthma during the 13-month study period September 1989 to September 1990, 2809 (95%) of which occurred among those aged 65 or under. Admission rates were highest in autumn, notably in September, and lowest in summer.

PM$_{10}$ values were obtained from a residential site in a wood-burning area.

The counts of asthma visits per day approximately followed a Poisson distribution and so Poisson regression was used in the analysis. In order to take account of autocorrelation in the number of daily visits the GEE approach of Liang and Zeger (1986) was used, in which the covariance structure is taken into account along with the variance in estimating regression coefficients. In addition, extra-Poisson variation was taken into account which tends to inflate the standard errors of the regression coefficients. As the effect of weather conditions was likely to be non-linear, dummy indicator variables for six ranges were constructed, for the minimum daily temperature on the previous days. Indicator variables for season, age stratum, day of the week and a variable for continuous time trend were also included in the regression model. As the lag whereby PM$_{10}$ may influence asthma visit is unknown, concentrations
lagged by up to several days previously as well as various multiple day averages were considered in analysis.

There was no association between ERVs in the elderly and PM$_{10}$ and so analyses were confined to those aged 65 or less. In the regression model controlling for previous day’s temperature categories, seasons, day of the week, hospital, September, time trends and age stratum, and allowing also for serial correlation and overdispersion, various indices of previous days’ PM$_{10}$ were significantly related to ERVs for asthma. The regression coefficient was 0.00367 (se 0.00126) per µg/m$^3$ increase in PM$_{10}$, considered as mean PM$_{10}$ of the previous four days.

As a Poisson model was used this can be interpreted as a percentage increase giving estimates of:

<table>
<thead>
<tr>
<th></th>
<th>Low</th>
<th>Mid</th>
<th>High</th>
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<tbody>
<tr>
<td></td>
<td>0.241%</td>
<td>0.367%</td>
<td>0.494%</td>
</tr>
<tr>
<td>per µg/m$^3$ PM$_{10}$</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

This estimate was similar in warm and in cold months; and was insensitive to various changes in how the effects of weather were represented. Likewise, omission of a high-pollen month had little effect on estimated percentage increases.

Though the underlying analyses refer only to those aged 65y or less, we have applied the results to people of all ages. (There is no logical reason why older people should be unaffected; and the study had little power to detect relationships in older people, there being only 146 ERVs for asthma over the 13-month period in those aged over 65.)

As Schwartz et al (1993) did not give background rates or the population data whereby they might be constructed, these need to be obtained elsewhere. Bates et al (1990) looked at attendance at emergency departments of nine acute care hospitals in Vancouver, British Columbia, Canada. The population was about 983,900 persons of all ages. Over the period 1 July 1984 to 30 June 1985 the number of visits for asthma was 3,440 giving a rate of 350 ERVs for asthma per year per 100,000 at risk. Applying the percentage increase from Schwartz et al (1993) to the baseline rate of Bates et al (1990) implies an

<table>
<thead>
<tr>
<th>Increase in ERV for asthma per 100,000 per year</th>
<th>Low 0.84</th>
<th>Change in</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mid 1.29</td>
<td>PM$_{10}$ (µg/m$^3$)</td>
</tr>
<tr>
<td></td>
<td>High 1.73</td>
<td></td>
</tr>
</tbody>
</table>

(Note/Erratum: because of a mistake in the background rates used, the values used in the ExternE implementations where $\frac{1}{2}$ of the figures given above and so are wrong.)
Schwartz et al (1991) conducted a longitudinal study of children’s visits with croup or obstructive bronchitis to children’s hospitals, to paediatric departments of general hospitals, or to paediatricians in five German cities (Duisburg, Köln, Stuttgart, Tübingen and Freudenstadt) during the mid 1980s. A diagnosis of croup was defined as acute stenotic subglottic laryngotraechitis, typical symptoms being hoarseness and barking cough, inspiratory stridor, dyspnoea, and with a sudden onset. There may have been difficulties in implementing the diagnostic criteria uniformly across many facilities. Mostly pre-school children are affected, with number of cases of croup peaking at two years of age.

Daily concentrations (µg/m$^3$) of SO$_2$, of TSP and of NO$_x$ were calculated using measurements from typically 2-4 monitors per city, for a period of 2-3 years. The median of the daily concentrations of TSP was about 50 µg/m$^3$ in Duisburg and Koblenz in Northrhine-Westfalia, and about 20 µg/m$^3$ in the other three (Southern German) cities. Median values of NO$_x$ were 14 µg/m$^3$ in one rural area in South Germany, and between 40-55 µg/m$^3$ in the other four cities. Poisson regression methods were used to examine, for each city separately and overall, the relationship between daily number of cases and (unlagged and lagged) daily measures of pollutants, adjusting first for seasonal patterns, weather and other temporal factors.

Daily cases of obstructive bronchitis were unrelated to daily changes in any of the three pollutants. Adjusting for other factors, daily cases of croup were related separately to all three pollutants, the strongest relationship being with TSP, the weakest with SO$_2$, though a relationship with NO$_x$ was also well established. The relationship with TSP was stronger on the log scale than on the ordinary scale, i.e. the relative impact of a 1 µg/m$^3$ increase was greater when background levels were lower. The overall Poisson regression coefficient 0.1244 (SE 0.0309) is equivalent to a central estimate of 0.124% increase in childhood croup per 1% increase in TSP with associated lower and upper values of 0.093% and 0.159% respectively.

We use baseline data from three Southern German cities, i.e. near where the Lauffen power plant is situated. There, 1% increase in TSP was equivalent to 0.2 µg/m$^3$. Also, on average there were 3.2 cases of croup per day in the study in the three Southern German cities (total population about 900,000 inhabitants); and study participation by hospitals and paediatricians was about 50%. This gives about 260 cases per year per 100,000 people of all ages at risk (the cases being in pre-school children). Assuming linearity, we derive the exposure-response relation:

<table>
<thead>
<tr>
<th>Additional cases of childhood croup per year, per 100,000 all ages population at risk</th>
<th>1.6 per µg/m$^3$ TSP</th>
</tr>
</thead>
</table>

Public Health Effects

with lower and upper estimates of 1.2 and 2.1 respectively. Using the conversion factors of Dockery and Pope (1994) (which of course may apply only approximately to Germany), we divide these estimates by 0.55 to give:

<table>
<thead>
<tr>
<th>Additional cases of childhood croup per year, per 100,000</th>
<th>Low 2.18 per µg/m³ PM₁₀</th>
<th>Med 2.91</th>
<th>High 3.82</th>
</tr>
</thead>
<tbody>
<tr>
<td>all ages population at risk</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>


Cody et al (1992) considered the relationship between emergency room visits and ambient ozone in the ‘ozone season’ (May to August) in 1988 and 1989 in central and northern New Jersey. Data on admissions for asthma and for bronchitis were obtained from nine hospitals, asthma being defined as ICD-9 codes of 493.9, 493.90 and 493.91. The total number of emergency visits at the nine hospitals in the study period was 147,000 with 814 classified as asthma and 912 as bronchitis. The mean age of asthma patients was 29 (SD=20) in 1988 and 30 (SD=20) in 1989.

Hourly ozone and SO₂ levels were measured in the five monitoring sites in central and northern New Jersey closest to the area serviced by the hospitals. The average ozone value between the hours of 10.00 to 15.00 was used as the primary exposure variable, and a mean ozone value was calculated across all five sites as a representative concentration for the region. There was a very high correlation, of 0.97, between the 5-hr index and 1-hr daily maximum of ozone. PM₁₀ data were available for every sixth day only.

The distribution of daily asthma visits was slightly positively skewed. Normal and Poisson distributions were assessed for their goodness-of-fit, and as a better fit was provided by the Normal distribution OLS regression was used to examine the relationship between ozone and daily ERVs for asthma. Lagged O₃ (of 24 and 48 hours) was considered as well as same-day ozone in the regression modelling, along with temperature, temperature change, 24-hr mean sulphur dioxide, relative humidity and visibility at noon. It would appear that no adjustment was made for possibly non-linear effects of temperature; and that the basis for adjustment for particles was poor.

Analyses were carried out both for the years 1988 and 1989 separately and on the combined dataset. There was a highly significant relationship between temperature and ERVs for asthma in all three sets of analyses. Adjusting for temperature, there was a statistically significant effect of 24-hr lagged ozone (24L) and/or same-day ozone, but not 48L O₃, in all analyses. The sparse PM₁₀ data showed no additional effect. (Ozone was unrelated to ERVs for bronchitis, where a weak association with PM₁₀ was found.) Omitting weekends, or adjusting for autocorrelation, made little difference to the results.

Detailed results varied somewhat by dataset, in terms both of ‘best model’ and of associated ozone effects. Estimated ozone effects from within-year analyses were slightly higher than in the pooled 1988-89 data, possibly suggesting some unadjusted confounding. We have chosen results from 1988 as arguably most representative of all the results presented and also the
most highly significant statistically. The regression coefficient was 0.0246 with standard error 0.0084 per ppb of ozone, 10.00 am to 3.00 ppm average concentration, lagged 24 hours.

The mean daily ERV for asthma was 3.3 during the 1988 study period. Applying this to the regression coefficient implies a (0.0246/3.3) = 0.75% increase in ERV per ppb ozone with low and high estimates of 0.49% and 1.00% respectively. (As usual, high and low estimates are given by one standard error above and below the mid-estimate.)

Because population data were not given in Cody et al (1992), and because that study referred to May-August only, baseline rates were taken from Bates et al (1990). The annual rate of emergency room visits for asthma (Bates et al 1990) was (3440/983,900) = 350 per 100,000 per year (see above). Applying the percentage increase from Cody et al (1992) to the baseline rate from Bates et al (1990) implies:

<table>
<thead>
<tr>
<th>Increase in ERV for asthma per 100,000 per year</th>
<th>Low 1.72</th>
<th>Change in Annual</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mid 2.63</td>
<td></td>
</tr>
<tr>
<td></td>
<td>High 3.50</td>
<td>O₃ (ppb)</td>
</tr>
</tbody>
</table>

where the exposure index is based on a specified 5-hr period within each day. Because of the very high correlation between that 5-hr index and 1-hr daily max. O₃, the strength of relationship with 1-hr max. ozone would have been very similar. The actual coefficients would however have depended on the conversion factor between the two indices. This is not given by Cody et al for the New Jersey data. In Germany however, the two indices are very similar (Wolfram Krewitt, personal communication) and so the above estimates were used, linked with 1-hr daily max. incremental ozone pollution, in the European implementations.

4.6.4 Restricted activity days (RADs)

(a) Strategy; and choice of source papers

One measure of morbidity, important in economic valuation though relatively rarely used in epidemiology, is a restricted activity day (RAD). A RAD is defined as a day when a study subject was forced to alter his/her normal activity. A severe restriction would include days when it was necessary to stay in bed. For employed adults, RADs include Work Loss Days (WLD); for children, days off school (whether or not the subject was confined to bed). Minor Restricted Activity Days (MRADs) do not involve work loss or bed disability.

These various measures consider the severity of the RAD. Some studies also investigate cause or ‘diagnosis’; for example, Respiratory-Related Restricted Activity Days (RRAD) might be considered especially relevant in the present context. Note that RADs are the most generalised outcome.

The principal available source of information of RADs and air pollution is the US Health interview Survey (HIS). The HIS is conducted annually by the National Centre for Health Statistics on a multistage probability sample of all adults aged 18-65 in 50,000 households from metropolitan areas of all sizes and regions throughout the USA. Data on social, economic, demographic and health status variables are collected by interview, either with the
study subject or with a close family member. In particular, there is a focus on acute illness in the two weeks prior to interview.

The HIS is a cross-sectional rather than a longitudinal study; i.e., each subject provides only one interview in any given year. This means that, unlike the other epidemiological studies from which E-R relationships for acute effects are derived, it is necessary to adjust in the analyses for socio-economic factors (e.g., age, race, sex, income, education).

Several analyses are available using HIS data from years 1976-81 inclusive to examine relationships between RADs and air pollution. Early papers, e.g., by Ostro (1983) studying 1976 data, were followed and superseded by several later papers including Portney and Mullahy (1986), Ostro (1987) and Ostro and Rothschild (1989). These papers focus on the number of relevant RADs in the two-week period prior to interview. Portney and Mullahy consider data from 1979 only, because supplemental data were available on the smoking habits and residence of a subsample of the 79,743 adults studied. Relationships were examined both with ozone and with sulphates, only those with ozone being statistically significant. Both Ostro (1987) and Ostro and Rothschild (1989) considered each of the six years 1976-81. Ostro (1987) considers only fine particles (FP, i.e., PM$_{2.5}$) estimated from airport visibility data, but examines adults without restriction. Ostro and Rothschild (1987) considered both FP and ozone, but only for adults who were in employment at time of interview, to ensure more reliable identification of RADs: workers’ days are more structured making restriction easier to define and recall. Also, workers’ activity patterns are more similar, and length of exposure to outdoor air pollutants are similar, than the population generally.

Both papers examining data from 1976-81 inclusive found important year-by-year differences in results; and in particular, there were indications that 1979 results may have been atypical. We therefore decided not to base quantification on Portney and Mullahy (1986). Relationships with ozone are not available from Ostro (1987); hence, Ostro and Rothschild (1989) was used. However, we used Ostro (1987) for E-R relationships with particles, on the grounds that, relative to the later paper, the greater representativeness of its population more than compensates for limitations in the determination of number of RADs.

(b) RADs and Particles: Ostro (1987)

Ostro (1987) used morbidity data collected from the total population sampled in the HIS for the years 1976 to 1981. Both WLDs and RADs were studied, in separate analyses, for each of these six years. Although WLDs appeared to be related to fine particles (FP; PM$_{2.5}$), there were major year-by-year differences in the estimated coefficients. Results for RADs showed a more consistent relationship with FP; these are used for the E-R relationships in the present study and are based on about 12,000 subjects per year.

Poisson regression methods were used, the dependent variable being the number of RADs per subject in a two-week period. The majority (85-95%) of subjects reported no restricted activity days. A fixed effects model was used by considering the deviation of individual observations from their city means; the paper lists 68 metropolitan areas included. In this
way, it was hoped to remove city-wide factors, such as time spent out of doors, building construction, and health practices. Air pollution was included as the relevant two week average of particulate matter, estimated from airport visibility data as FP (PM$_{2.5}$). Other variables added to the regression model included age, sex, race, education, income, quarter of survey, marital status, existence of a chronic condition, and average two week minimum temperature. A variable was also added to indicate whether the individual was working or not.

Because a Poisson regression model was used, the regression coefficients can be interpreted as percentage changes per µg/m$^3$ increases in two week average FP. The six coefficients for RAD as outcome showed substantial year-by-year variation, ranging from 0.284 to 0.900; though each was highly significant statistically (p<0.01). Averaging across years, the geometric mean regression coefficient was 0.438 per µg/m$^3$ FP. Using the conversion factor (FP=) PM$_{2.5}$ = 0.6 PM$_{10}$ (Dockery and Pope, 1994), this implies percentage changes of

<table>
<thead>
<tr>
<th>Level</th>
<th>Percentage Change per µg/m$^3$ PM$_{10}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low</td>
<td>0.167 %</td>
</tr>
<tr>
<td>Mid</td>
<td>0.263 %</td>
</tr>
<tr>
<td>High</td>
<td>0.412 %</td>
</tr>
</tbody>
</table>

The US Coal Fuel Cycle Report (ORNL/RFF, 1994) gives a background rate of 19 RADs per person per year, equivalent to a prevalence of 5.2%. Linking this background rate with the above percentage increases gives the following annual increases:

<table>
<thead>
<tr>
<th>Increase in RADs per 1,000 adults per year</th>
<th>Low 31.8</th>
<th>Mid 49.9</th>
<th>High 78.3</th>
<th>Change in PM$_{10}$ (µg/m$^3$)</th>
</tr>
</thead>
</table>

(c) Minor Restricted Activity Days (MRADs) and Ozone: Ostro and Rothschild (1989)

Ostro and Rothschild (1989) considered the same six years of the HIS (1976-1981), and focused on minor (MRADs) and respiratory (RRADs) activity days. Only current workers, resident in urban areas, were included.

A Poisson multiplier regression model was used, as in the earlier analysis (Ostro 1987), in separate year-by-year analyses of the data. Two-week averages of the daily ozone levels (daily 1-hr max., in µg/m$^3$) were used in the analysis. The regression coefficients for ozone were also adjusted for FP as well as various socio-economic confounders. Perhaps surprisingly, there was no clear or consistent relationship linking ozone and RRADs. There was however a reasonably strong and consistent relationship between MRADs and ozone. The regression coefficients for the six years were again very variable with most (including two negative!) being statistically significant individually. The highest and lowest values were discarded to give an (arithmetic) mean regression coefficient from the other four years of 0.085 (SE = 0.2) per µg/m$^3$ of ozone. This is equivalent to 0.200 (SE = 0.470) per ppb ozone. (1 ppb = 2.3 µg/m$^3$ O$_3$).
As a Poisson regression model was used the regression coefficients can be interpreted as a percentage increase per ppb ozone. Thus, percentage increases in MRADS are

<table>
<thead>
<tr>
<th></th>
<th>Low 0.270 %</th>
<th>Mid 0.200 %</th>
<th>High 0.670 %</th>
</tr>
</thead>
</table>

The low figure was assumed to be zero, since it is unlikely that increases in ozone would lead to a reduction in MRADS. The data from Ostro (1987) implied a baseline value for MRAD of approximately 15 days per year. However, Ostro and Rothschild (1989) gave a mean MRAD of 7.8 days per year, so this was used in the calculations below. Applying the percentage increase above to the mean MRAD = 7.8 days per year gave the following results.

<table>
<thead>
<tr>
<th>Increase in MRADS per 1000 population</th>
<th>Low 0</th>
<th>Change in Annual O&lt;sub&gt;3&lt;/sub&gt; per year</th>
</tr>
</thead>
<tbody>
<tr>
<td>per year</td>
<td>Mid 15.6</td>
<td>High 52.3</td>
</tr>
</tbody>
</table>

### 4.6.5 Provocation or exacerbation of asthma

**a) Choice of endpoint and of papers**

There is a difficulty with this endpoint, because there are differences between studies in the definitions both of ‘asthmatic’ and of ‘asthma attacks’. The two principal papers available at time of review to quantify the acute effects of PM pollution on asthmatics were Ostro et al (1991) and Pope et al (1991). We chose Ostro et al because of the larger number of subjects studied; and because of endpoint studied. This in effect was ‘shortness of breath’ in asthmatics, which clearly is a response variable; i.e., plausibly, a consequence of pollution. Pope et al include much fewer subjects; and focus on lung function, which ExternE cannot yet cost; and on bronchodilator usage.

A case can be made for treating bronchodilator usage as a health-related endpoint in its own right, as well as quantifying ‘asthma attacks’. This should be reviewed in any further phase of the project.

**b) Particles and shortness of breath in asthmatics: Ostro et al (1991)**

Ostro et al (1991) considered the effects of daily concentrations of particles including sulphates, nitrates and PM<sub>2.5</sub> on adult asthmatics in Denver. The 207 study subjects were predominantly white, female, employed and well-educated; and generally with moderate to severe asthma. Participants in the study recorded daily information throughout the winter of 1987 to 1988 on the presence and severity (measured on a scale from 0 to 4) of cough, wheeze, shortness of breath, chest tightness and sputum production, along with physician visits and emergency room visits.
The ambient air pollutants studied were daily measures of sulphates, nitrates, PM$_{2.5}$, nitric acid, hydrogen ion (H$^+$) and sulphur dioxide. Daytime average (9.00 am to 4.00 pm) data were used. These were highly correlated with 24-hr. average data; conversion factors not given. Furthermore, an attempt was made to derive individual estimates of ‘dose’, by adjusting the ambient pollution level with duration of exposure, the time spent indoors or outdoors, and ventilation rates based on exercise. The relationship between ambient pollution and these derived variables is not given; hence there is some ambiguity about the scale of measurement of pollutants.

The results reported were based on linear regression methods, with outcome variable the probability of reporting the symptom of interest on any given day. These analyses included adjustment for day of survey, symptom on previous day, an indicator for weekday or weekend, use of gas stove, minimum temperature and autocorrelation; though not for other pollutants, which were included separately and one-at-a-time into the regression modelling. Ozone levels were low in this winter-time study and therefore were dismissed as a potential confounding factor. Logistic regression and other sensitivity analyses gave similar results.

The principal relationships were found with measures of hydrogen ion (H$^+$). It was found that the reporting of moderate or worse shortness of breath was also significantly associated with daily sulphate levels. The results suggest an increase in shortness of breath days = 0.0077 (SE = 0.0038) per unit SO$_4$ µg/m$^3$ on the logarithmic scale (Ostro et al, Table 5); i.e., a result just statistically significant at the 5% level. ‘Asthma’ and ‘cough’, as used in the paper, were not well related to ambient particulate pollution; and indeed ‘shortness of breath’ in asthmatics, though strongly related to sulphates, was not identified as related to PM$_{2.5}$; a somewhat surprising result, given the high correlation (r=0.88) between daily (24-hr average) ambient sulphates and PM$_{2.5}$.

Despite these ambiguities in results, and lack of information about how daytime average sulphate values correspond to 24-hr average values as used in ExternE, we have used the relationship with sulphates in quantifying effects. The principal relationship given can be linearised at either the mean for the study (2.11 µg/m$^3$) or at some higher value. Linearising at a low level for a relationship with exposure on the logarithmic scale would give a steeper slope than linearising at a higher value; and hence would over-estimate the effects when applied at higher background levels of pollution. We linearised the slope at the somewhat arbitrary value of 5 µg/m$^3$ daytime average SO$_4$, approximately equivalent to 20 µg/m$^3$ PM$_{10}$ if the conversion factors used by Dockery and Pope (1994) apply. This value is closer to that needed for applications in Europe based on 24-hr average data, though still untypically low for many European locations.

Using these approximations, linearising at 5 µg/m$^3$ sulphates, assuming that PM$_{10}$ includes 25% sulphates (Dockery and Pope, 1994) and annualising gives an increase in shortness of breath days per year per asthmatic = 0.0077/20 x 365 = 0.14. The full results are given below.

<table>
<thead>
<tr>
<th>Increase in Shortness of breath days per year per asthmatic</th>
<th>Low 0.07</th>
<th>Change in annual PM$_{10}$</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>*</td>
<td>(µg/m$^3$)</td>
</tr>
<tr>
<td></td>
<td>Mid 0.14</td>
<td></td>
</tr>
<tr>
<td></td>
<td>High 0.21</td>
<td></td>
</tr>
</tbody>
</table>
c) Ozone and ‘asthma attacks’: Holguin et al (1985)

As noted earlier (4.3.5(b)), Holguin et al (1985) is a summertime (May to October, 1981) study of 42 young (median age 13y, range 7-55y) non-smoking asthmatics of moderate severity, near Houston, Texas, USA, where ‘the composition of summertime photochemical pollution may be quite different from other well-studied areas’. Results refer to 12-hour periods (‘daytime’, 7am-7pm; and ‘night-time’, 7pm to 7am). Asthmatic episodes were defined for each subject by occurrence of symptoms, decreasing peak expiratory flow (PEF) and increased use of medication.

Ozone and NO₂ were each represented by the maximum 1-hr concentration in the relevant 12-hr period. These are derived estimates of individual exposures, and not ambient local concentrations only; an advantage in attempting to identify relationships, but a difficulty in using results within the ExternE project. Particles were measured but no results reported; presumably no relationships found.

Subject-specific logistic regression analyses included attack status in the previous 12-hour period, temperature, relative humidity, pollen count and NO₂ (which was correlated with O₃). In this context, O₃ was positively related to asthma attacks for most subjects. Interactions (e.g., of O₃ with previous attack) were not examined; nor were possible dependencies on attack status earlier than previous period. Lagged exposure variables were not included.

A random coefficients model was used to combine, across subjects, O₃ coefficients from the logistic regression models. There was a statistically significant relationship between ozone level and the probability of an asthma attack in the 12-hr periods, with a regression coefficient (log odds) of $6.20 \times 10^{-3}$ per O₃ (ppb) with SE = $2.31 \times 10^{-3}$. This result plus and minus one standard error is

<table>
<thead>
<tr>
<th>Low</th>
<th>0.00389</th>
</tr>
</thead>
<tbody>
<tr>
<td>Med.</td>
<td>0.00620</td>
</tr>
<tr>
<td>High</td>
<td>0.00851</td>
</tr>
</tbody>
</table>

per ppb ozone

Some work is needed to re-express this result in a usable form for application in ExternE.

i. The baseline attack rate was 15% per 12-hour period. This is the stationary probability of attack $P_i = p_i / (1 - p_i + p_o)$, as described in Krupnick et al (1990; see 4.6.6 below) where

$p_i =$ mean transition probability (illness day t given illness on day t-1) and

$p_o =$ mean transition probability (illness on day t given no illness on day t-1).

Also, the odds ratio for a new attack given following an occurrence in the previous period, compared with when no attack occurred in the previous period, was $\exp(1.22) = 3.39$. Together, these two pieces of information imply values of $p_o = 0.12$ and $p_i = 0.32$.  

129
ii. Following standard theory, as described for example by Krupnick et al (1990), the appropriate estimate of health effects per increment of pollution is given by the derivative of the stationary probability with respect to the pollutant of interest. Following Krupnick et al (1990) this simplifies to

\[ \pi' = \beta \frac{p_0(1 - p_1)(1 - p_0 + p_1)}{(1 - p_1 + p_0)^2} \]  

or

\[ \pi' = \beta \lambda \]

Substituting the mean stationary probabilities as \( p_0 = 0.12 \) and \( p_1 = 0.32 \) gives \( \lambda = 0.153 \). Applying this to the regression coefficients gives an increase in stationary probability of attack in 12-hour period of

<table>
<thead>
<tr>
<th></th>
<th>Low</th>
<th>Med.</th>
<th>High</th>
</tr>
</thead>
<tbody>
<tr>
<td>( \pi' )</td>
<td>0.000595</td>
<td>0.000949</td>
<td>0.001302</td>
</tr>
</tbody>
</table>

per ppb ozone

i. The estimate that \( p_1 = 0.32 \) can also be used to derive an attack rate for a 24-hr, not 12-hr, period. If, on average, 15% of subjects have an attack in a given 12-hr period, then on average 0.32 of these, or 4.8% of all subjects, have a recurrence in the following 12-hour period; i.e., on average, about 10% of subjects have a new attack in the subsequent 12-hr period. It follows that on average about 25% of subjects experienced an attack in any 24-hr period, some of these subjects experiencing more than one attack in that time; i.e., the estimates of (ii) above need to be inflated by a factor of 1.68 to give increases in attacks per 24-hr period per ppb O₃.

Multiplying by 365 to annualise, it follows that:

<table>
<thead>
<tr>
<th>Annual number of asthma attack days =</th>
<th>Low 0.365</th>
<th>Mid 0.582</th>
<th>High 0.798</th>
</tr>
</thead>
<tbody>
<tr>
<td>Change in annual O₃ (ppb)</td>
<td>( \ast )</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

4.6.6 (Respiratory) symptoms

(a) Choice of paper

Krupnick et al (1990) consider presence of any one of several symptoms; Ostro et al (1993) consider incidence of episodes of unknown duration. There are advantages, from the viewpoint of economic valuation in terms of symptom-days, in using presence/absence data; though analyses by Schwartz and co-workers of the Los Angeles Student Nurses dataset have
shown that the air pollutants which influence the risk of occurrence (incidence) of a symptoms episode may differ from those that influence the duration of an episode. Also, Ostro et al (1993) restricted their analyses to non-smokers; results may be less representative. Hence we use Krupnick et al (1990).

(b) Particles, ozone and symptom days: Krupnick et al (1990)

i. The study: Krupnick et al (1990) conducted a large panel study of 290 families (572 adults, 756 children) in California from September 1978 to March 1979. About 70% of the adults studied were aged 30-45; i.e. older people and younger adults were seriously under-represented, compared with the general population. The pollutants considered were daily exposure to ozone (daily 1-hr max.), the coefficient of haze (COH) as a surrogate for fine particles (daily average), sulphur dioxide (daily average) and nitrogen dioxide average of peak period).

Health effects were measured as the presence or absence of any one of 19 respiratory related symptoms or conditions; headache, and eye irritation. Two binary outcomes were derived for analysis, namely the presence of a serious respiratory symptom and the presence of any symptom or condition, irrespective of severity. This latter, more general, outcome is considered here. From the viewpoint of economic valuation it is noteworthy that about 1 in 24 (i.e. <5%) of these symptoms were classed as ‘serious’, where serious is interpreted as a Restricted Activity Day or presence of fever or ‘sought medical advice.’

As there was no significant effect of ozone in children and the relationships for COH were similar in adults and children, the results for adults were used to derive exposure-response relationships for all ages.

A Markov process model was used to analyse the relationship between air pollution and these (mild) symptoms, thus taking into account that the probability of illness on any day is likely to be dependent on the occurrence or not of symptoms on the previous day. Logistic regression was used to model the probability of symptoms on any given day, with adjustment for the dependent variable lagged one day as one of several explanatory variables.

The design was of daily pooled cross-sections over a longitudinal follow-up. Thus, possible confounding included both factors that varied day-by-day (temperature, rainfall humidity) but also a wide range of characteristics at the individual level. A major strength of the study was the use made of a very large number (74682) person-days of observations on adults. This enabled careful modelling of different combinations of pollutants, while adjusting for numerous other covariates.

Modelling suggested an important role both of particles (COH) and of ozone in relation to presence or absence of symptoms. Very marked but curious effects were found when several pollutants (COH, ozone, NO₂, SO₂) were included simultaneously in regression models. These effects are a reminder of the complexities involved when looking to
attribute effects to the various components of the air pollution mixture. It is rare to have these difficulties demonstrated so clearly.

ii. Particles: The relationship between COH and the probability of reporting one of the symptoms is summarised by the logistic regression coefficient (log odds) of 0.0088 (SE = 0.0046) / 100 ft, adjusted for ozone and sulphur dioxide (though inclusion or not of SO₂ was in fact irrelevant. Adjustment for ozone did matter.) Using the conversion factor COH = 0.55 PM₁₀ (Dockery and Pope, 1994), this result implies a mid-estimate of log-odds of change in symptom days as 0.0088 x 0.55 = 0.00484. The baseline probability of any (adult) subject reporting any symptom is given as 0.19. This is therefore the stationary probability described also as \( p_1 = p_0 / (1 - p_1 + p_0) \) where

\[ p_1 = \text{mean transition probability (illness day } t \text{ given illness on day } t-1) \text{ and} \]

\[ p_0 = \text{mean transition probability (illness on day } t \text{ given no illness on day } t-1) \]

The mean values of \( p_i \) and \( p_o \) were not stated in the paper. However, knowledge that the baseline (stationary) probability is 0.19 overall, and that the log odds of \( p_i \) relative to \( p_o \) is 4.065, allows these to be estimated as \( p_o = 0.0550 \) and \( p_i = 0.7655 \). (Hagler/Bailly, 1995, report a personal communication with Krupnick et al, 1990, that the mean adult values were \( p_i = 0.7775 \) and \( p_0 = 0.0468 \); i.e., the estimates we used were similar.)

As before (discussion of Holguin et al, 1985, above), the incremental effect of pollution is given by the derivative of the stationary probability with respect to the pollution variable is the appropriate health effect measure. Then the health effect is given by

\[ \pi = \beta \lambda \]

where, after substitution for \( p_0 \) and \( p_1 \), \( l = 0.2632 \).

Having obtained a value for \( l = 0.2632 \), the estimated annual effect of PM₁₀ for all ages is given as annual change in symptom days per 1000 people at risk (all ages) = 0.00484 x 0.2632 x 365 x 1000 = 465.0 per change in PM₁₀ (µg/m³). The full results are given below.

<table>
<thead>
<tr>
<th>Annual change in Symptom days per 1000 people at risk</th>
<th>Low</th>
<th>221.9</th>
<th>Change in PM₁₀ (µg/m³)</th>
</tr>
</thead>
<tbody>
<tr>
<td>all ages)</td>
<td>Mid</td>
<td>465.0</td>
<td>PM₁₀</td>
</tr>
<tr>
<td></td>
<td>High</td>
<td>686.9</td>
<td>(µg/m³)</td>
</tr>
</tbody>
</table>

iii. Ozone: Results for ozone are taken from the same paper, same statistical model. In particular, the ozone effects are adjusted for particles. The key relationship is summarised by the regression coefficient 0.0055 (SE = 0.0027) per O₃ (ppm), which is equivalent to 0.00055 (SE = 0.00027) per O₃ (ppb).

The results were not applied to children as no significant relationship was found for children.
As with particles it is necessary to take into account the probability of illness on day $t$ conditional on whether illness was present in the previous day, $t-1$. Using the same reasoning as for particles the health effect of ozone is given by $0.00055 \times 0.2632 = 0.0001448$. The full results are given below.

<table>
<thead>
<tr>
<th>Annual change in Symptom days in adults per 1000 adults at risk</th>
<th>Low 26.9</th>
<th>Mid 52.8</th>
<th>High 78.8</th>
</tr>
</thead>
</table>

### 4.7 Exposure-response (E-R) Relationships: Chronic Health Effects

#### 4.7.1 Mortality

Pope et al (1995c) looked at mortality in a large prospective study of 500,000 individuals who had been followed-up over 8 years. These individuals lived in 151 UK metropolitan areas and ambient concentrations of sulphates and fine particles ($\text{PM}_{2.5}$, aerodynamic diameters less than or equal to 2.5 mm) were obtained from 50 metropolitan areas. As sulphates and $\text{PM}_{2.5}$ are products of fossil fuels and sulphates make up the largest proportion of fine particles by mass, they were highly correlated ($r = 0.73$). The mean sulphate level was 11.0 (sd = 3.6) with a range of 3.6 to 23.5 $\mu g/m^3$, while the mean $\text{PM}_{2.5}$ level was 18.2 (sd = 5.1) with a range of 9.0 to 33.5 $\mu g/m^3$. Sulphates were significantly associated with all cause mortality with RR = 1.15 (95% CI = 1.09, 1.22) for increases of 19.9 $\mu g/m^3$ SO$_4$. Fine particles ($\text{PM}_{2.5}$) were also significantly associated with all cause mortality with RR = 1.17 (95% CI = 1.09, 1.26) for increases of 24.5 $\mu g/m^3$ PM$_{2.5}$. Both these results were adjusted for age, sex, race, cigarette smoking, exposure to passive cigarette smoke, body mass index, drinks per day of alcohol, education and occupational exposure. The result for $\text{PM}_{2.5}$ of RR = 1.17 (1.09, 1.26) is equivalent to an increase of 40.8 $\mu g/m^3$ of PM$_{10}$.

This implies RR $= 1.00386$ per $\mu g/m^3$ PM$_{10}$ or a 0.386% increase in mortality per $\mu g/m^3$ PM$_{10}$. With a standard error of 0.037 for the regression coefficient, the following results were obtained:

<table>
<thead>
<tr>
<th>Percent increase in chronic mortality</th>
<th>Low 0.295 %</th>
<th>Mid 0.386 %</th>
<th>High 0.477 %</th>
</tr>
</thead>
</table>

#### 4.7.2 Respiratory morbidity

(a) Bronchitis and chest illness in adults: Schwartz (1993b)

Schwartz (1993b) looked at the association between chronic respiratory disease and particulate air pollution using data from the first National Health and Nutrition Examination Survey (NHANES I), conducted between 1971 and 1975. Analyses were based on 3874 subjects aged 30-74 years from 53 urban areas in the USA. Thus the NHANES I survey
covers a wide section of the USA, both urban and rural, giving some safeguards against the accidental confounding of air pollution with other factors.

These subjects had answered a detailed medical history questionnaire, including questions on chronic respiratory disease and lifetime smoking habits. The outcomes of chronic bronchitis and asthma were defined by affirmative responses to both a doctor recall question ‘Has a doctor ever told you had (asthma/chronic bronchitis)?’ and the question ‘Do you still have it?’. Subjects who had affirmative doctor recall for asthma/chronic bronchitis but who reported that they no longer had the condition were excluded from the analysis. Respiratory illness was based on physician diagnosed ICD8 (460-519). Dyspnoea was defined as ‘shortness of breath when hurrying on a level or walking up a slight hill’.

Monitors from the NHANES I locations gave long-term TSP concentrations for the year preceding the physical examination and questionnaire. The mean TSP concentration for each location was obtained by first averaging all monitors on each day and then averaging all days in each year. Across study subjects, the median TSP value calculated in this way was 81 µg/m$^3$ (5th percentile 48 µg/m$^3$, 95th percentile 131 µg/m$^3$).

In the analysis the probability of having respiratory symptoms or a diagnosis of respiratory disease was modelled using logistic regression, adjusting for age, race, sex, and smoking. Models allowing for various parameterisations of smoking history were considered, and a baseline model developed including several smoking variables. TSP was then examined as a risk factor. Also, in order to examine the nature of the relationship between chronic respiratory conditions and TSP, indicator variables for quartiles of TSP were examined in the regression model (in place of the continuous TSP index).

The results showed that TSP was significantly associated with (partially self-reported) chronic bronchitis (OR = 1.07, 95% C.I.= 1.02, 1.12) and (physician diagnosed) respiratory illness (OR = 1.06, 95% C.I. = 1.02, 1.11), these being odds ratios for a 10 µg/m$^3$ increase in long term average TSP concentration. The odds ratios increased for the 1504 never smokers when analysed separately, suggesting that the association was not due to lack of control for smoking. There was no significant association between TSP air pollution and the prevalence of asthma or dyspnoea. The analysis using quartiles of TSP suggested a relationship which could acceptably be described as linear, with no evidence of a threshold, and some evidence of a ‘levelling off’ at higher average concentrations (although with a fair amount of unexplained scatter).

For chronic bronchitis, the quoted odds ratios and associated 95% CI are equivalent to a logistic regression coefficient of 0.00677 (SE 0.00234) per µg/m$^3$ TSP. Using the conversion factor PM$_{10}$ = 0.55 TSP, this implies a change in log odds for chronic bronchitis of (0.00677/0.55) = 0.0123 (SE 0.00425) per µg/m$^3$ PM$_{10}$ (annual average). For respiratory illness, using the same approach and conversion factor implies an increase in the log odds of 0.00583/0.55 = 0.1060 per 1 µg/m$^3$ of PM$_{10}$. Background prevalence rates for bronchitis (6%) and respiratory illness (10%) were estimated as the mean value for white subjects aged 30 to 74. (These values were higher than for non-whites.) Applying the incremental increases in
log odds to these background rates gives increases in the prevalence of chronic bronchitis and respiratory illness in adults as follows:

<table>
<thead>
<tr>
<th>Increase in prevalence</th>
<th>Low 45</th>
<th>Change in average PM$_{10}$</th>
<th>Mid 70</th>
<th>*</th>
<th>High 94</th>
<th>(µg/m$^3$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>chronic bronchitis per 100,000 adults per year</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>respiratory illness per 100,000 adults per year</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

(b) Bronchitis and chronic cough in children: Dockery et al (1989)

Dockery et al (1989) examined the relationships between particles and respiratory illness (1980-81) in a cohort of 5422 white school-children aged 10-12 years, from the six cities in Eastern United States participating in the Harvard Six-Cities study. Five respiratory outcomes were considered in a cross-sectional analysis: doctor’s diagnosis of bronchitis in the last year, chronic cough present for at least three months in the last year, chest illness which restricted activity for at least three days in the previous year, persistent wheeze and doctor diagnosis of asthma.

Daily mean concentrations of SO$_2$, NO$_2$ and O$_3$ were obtained by averaging hourly concentrations, for each day with at least 18 hourly values. Three measures of particles were available: TSP, PM$_{15}$ and PM$_{2.5}$. Monthly means were calculated for each pollutant by averaging all available daily values. Air pollution exposure in the previous year was calculated for each child by averaging monthly means for the 12 months preceding the examination.

A two step analysis was used to account for city to city variation in health outcomes. Initial logistic regressions modelled the probability of illness in each of the six cities separately, controlling for sex, age, maternal smoking and the presence of a gas stove in the home. In the next step, the estimated logits were regressed against city-specific air pollution measurements using weights inversely proportional to the sum of the between city variance and the within-city variance of the adjusted logits.

The results of the regression analyses were expressed as odds ratios of illness in the most polluted relative to the least polluted city. For PM$_{15}$, these values were a ‘low’ of 20.1 µg/m$^3$ in Portage and a ‘high’ of 58.8 µg/m$^3$ in Steubenville; i.e., a difference of 38.7 µg/m$^3$ PM$_{15}$.

Adjusting for non-pollution confounders, and re-expressing as odds ratios per unit exposure, results showed that asthma was statistically significantly related to O$_3$ but not to particles; and PM$_{15}$ (but not TSP or PM$_{2.5}$) was statistically significantly related to bronchitis (OR = 2.2, 95% CI 1.1, 4.2) and to chronic cough (OR = 4.1, 95% CI 1.9, 9.2). Additional separate analyses were carried out for children with a history of wheeze or asthma and those with no history,
showing that the prevalence of illness was higher and the associations with air pollution stronger in those with a history.

Re-expressing the odds ratios in terms of unit exposure gives, for bronchitis, $\text{OR} = 1.02$ (95% CI = 1.003, 1.048) and, for chronic cough, $\text{OR} = 1.03$ (95% CI = 1.0, 1.069), for 1 $\mu$g/m$^3$ increase of PM$_{10}$. Combining, as for Schwartz (1993) above, the background average (mean) rates of bronchitis (6.47%) and chronic cough (5.77%) among children studied in all six cities with the logistic regression coefficients implied by these odds ratios; converting from PM$_{15}$ to PM$_{10}$ using the factors PM$_{10} = 0.9$ PM$_{15}$; and taking, as always, ‘high’ and ‘low’ values as 1 s.e. from the mid-estimate gives the following annual increments per $\mu$g/m$^3$ of PM$_{10}$:

<table>
<thead>
<tr>
<th>Increase in prevalence of children with bronchitis per 100,000</th>
<th>Low 85</th>
<th>Change in annual (µg/m$^3$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Increase in prevalence of children with chronic cough per 100,000</td>
<td>Low 103</td>
<td>Change in annual (µg/m$^3$)</td>
</tr>
</tbody>
</table>

### 4.8 Summary of Exposure-Response Relationships Proposed

Acute effects quantified in the current phase of ExternE now include mortality, hospital admissions, (hospital) emergency room visits (ERVs), restricted activity days (RADs), asthma attacks and (mostly respiratory) symptoms. Lung function effects are not described because there is currently no good means of economic valuation. There is also a first quantification of the effects of particles on chronic respiratory disease. See earlier for a discussion of chronic mortality.

The relationships proposed are summarised in the following Tables 4.2 to 4.5.
Table 4.2 Exposure-response functions used for the assessment of mortality effects.

<table>
<thead>
<tr>
<th>Pollutant: PM$_{10}$</th>
<th>Percent change</th>
<th>Low</th>
<th>Mid</th>
<th>High</th>
<th>Change in annual mortality</th>
<th>Change in annual PM$_{10}$ concentration in [µg/m$^3$]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Schwartz (1993a)</td>
<td></td>
<td>0.064</td>
<td>0.104</td>
<td>0.145</td>
<td>=</td>
<td>*</td>
</tr>
<tr>
<td>in acute mortality</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pope et al (1995c) 1</td>
<td>Percent change</td>
<td>0.295</td>
<td>0.386</td>
<td>0.477</td>
<td>=</td>
<td></td>
</tr>
<tr>
<td>in chronic mortality</td>
<td>Low</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>in [µg/m$^3$]</td>
<td>Mid</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Pollutant: Ozone</th>
<th>Percent change</th>
<th>Low</th>
<th>Mid</th>
<th>High</th>
<th>Change in annual acute mortality</th>
<th>Change in annual O$_3$ concentration in [ppb]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Kinney et al (1994)</td>
<td></td>
<td>0.010</td>
<td>0.015</td>
<td>0.020</td>
<td>=</td>
<td></td>
</tr>
<tr>
<td>change in acute mortality</td>
<td>Mid</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

1 Note: Results for chronic mortality are not included in the summary tables of estimated effects and costs in the main fuel cycle reports.
Table 4.3 Exposure-response functions used for the assessment of acute morbidity effects due to incremental particulate air pollution.

<table>
<thead>
<tr>
<th>Source</th>
<th>Effect</th>
<th>Exposure Parameter</th>
<th>Exposure Value</th>
<th>Change in Annual Exposure</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Schwartz (1994d)</td>
<td>Change in hospital admissions</td>
<td>Low</td>
<td>0.124</td>
<td>Change in annual</td>
<td>PM$_{10}$ concentration</td>
</tr>
<tr>
<td>and</td>
<td>for respiratory infections, per</td>
<td>Mid</td>
<td>0.187</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Burnett et al (1994)</td>
<td>100000 persons (all ages) per yr</td>
<td>High</td>
<td>0.251</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Schwartz (1994d)</td>
<td>Change in hospital admissions</td>
<td>Low</td>
<td>0.161</td>
<td>Change in annual</td>
<td>PM$_{10}$ concentration</td>
</tr>
<tr>
<td>and</td>
<td>for COPD per 100000 persons</td>
<td>Mid</td>
<td>0.227</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Burnett et al (1994)</td>
<td>(all ages), per year</td>
<td>High</td>
<td>0.293</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sunyer et al (1993)</td>
<td>Change in ERVs for COPD, per 100000 persons (all ages), per year</td>
<td>Low</td>
<td>0.58</td>
<td>Change in annual</td>
<td>PM$_{10}$ concentration</td>
</tr>
<tr>
<td>Schwartz et al (1993)</td>
<td>Change in ERVs for asthma, per 100000 persons (all ages), per year</td>
<td>Low</td>
<td>0.84</td>
<td>Change in annual</td>
<td>PM$_{10}$ concentration</td>
</tr>
<tr>
<td>Bates et al (1990) #</td>
<td></td>
<td>Mid</td>
<td>1.29</td>
<td></td>
<td></td>
</tr>
<tr>
<td>High</td>
<td>1.73</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Schwartz et al (1991)</td>
<td>Change in hospital visits for childhood croup per 100000 persons (all ages), per year</td>
<td>Low</td>
<td>2.18</td>
<td>Change in annual</td>
<td>PM$_{10}$ concentration</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mid</td>
<td>2.91</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>High</td>
<td>3.82</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ostro (1987)</td>
<td>Change in RADs per 1000 adults per year</td>
<td>Low</td>
<td>31.8</td>
<td>Change in annual</td>
<td>PM$_{10}$ concentration</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mid</td>
<td>49.9</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>High</td>
<td>78.3</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ostro et al (1991)</td>
<td>Change in ‘shortness of breath’ days per asthmatic per year</td>
<td>Low</td>
<td>0.07</td>
<td>Change in annual</td>
<td>PM$_{10}$ concentration</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mid</td>
<td>0.14</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>High</td>
<td>0.21</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Krupnick et al (1990)</td>
<td>Change in symptom days, per 1000 persons (all ages), per year</td>
<td>Low</td>
<td>221.9</td>
<td>Change in annual</td>
<td>PM$_{10}$ concentration</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mid</td>
<td>465.0</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>High</td>
<td>686.9</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

COPD: Chronic Obstructive Pulmonary Disease.
ERV: Emergency Room Visit.
RAD: Restricted Activity Days.

# This value has been corrected since implementation; see text, area.
Table 4.4 Exposure-response functions used for the assessment of acute morbidity effects due to incremental ozone air pollution.

<table>
<thead>
<tr>
<th>Author(s)</th>
<th>Description</th>
<th>Level</th>
<th>Coefficient</th>
<th>Unit</th>
<th>Coefficient Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Schwartz (1994d)</td>
<td>Change in hospital admissions for respiratory infections, per 100000 persons (all ages), per year</td>
<td>Low</td>
<td>0.227</td>
<td></td>
<td>Change in annual</td>
</tr>
<tr>
<td>and Burnett et al (1994)</td>
<td></td>
<td>Mid</td>
<td>0.303</td>
<td>* O₃ concentration</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>High</td>
<td>0.361</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Schwartz (1994d)</td>
<td>Change in hospital admissions for COPD, per 100000 persons (all ages), per year</td>
<td>Low</td>
<td>0.138</td>
<td></td>
<td>Change in annual</td>
</tr>
<tr>
<td>and Burnett et al (1994)</td>
<td></td>
<td>Mid</td>
<td>0.220</td>
<td>* O₃ concentration</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>High</td>
<td>0.303</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Thurston et al (1994b)</td>
<td>Change in hospital admissions for asthma, per 100000 persons (all ages), per year</td>
<td>Low</td>
<td>0.283</td>
<td></td>
<td>Change in annual</td>
</tr>
<tr>
<td>and Burnett et al (1994)</td>
<td></td>
<td>Mid</td>
<td>0.571</td>
<td>* O₃ concentration</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>High</td>
<td>0.858</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cody et al (1992)</td>
<td>Change in ERVs for asthma, per 100000 persons (all ages), per year</td>
<td>Low</td>
<td>1.72</td>
<td></td>
<td>Change in annual</td>
</tr>
<tr>
<td>and Bates et al (1990)</td>
<td></td>
<td>Mid</td>
<td>2.63</td>
<td>* O₃ concentration</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>High</td>
<td>3.50</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ostro and Rothschild (1989)</td>
<td>Change in minor RADs, per 1000 adults, per year</td>
<td>Low</td>
<td>0</td>
<td></td>
<td>Change in annual</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mid</td>
<td>15.6</td>
<td>* O₃ concentration</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>High</td>
<td>52.3</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Holguin et al (1985)</td>
<td>Change in asthma attacks per asthmatic, per year</td>
<td>Low</td>
<td>0.365</td>
<td></td>
<td>Change in annual</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mid</td>
<td>0.582</td>
<td>* O₃ concentration</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>High</td>
<td>0.798</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Krupnick et al (1990)</td>
<td>Change in symptom days, per 1000 persons (all ages), per year</td>
<td>Low</td>
<td>26.9</td>
<td></td>
<td>Change in annual</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mid</td>
<td>52.8</td>
<td>* O₃ concentration</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>High</td>
<td>78.8</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

COPD: Chronic Obstructive Pulmonary Disease.
ERV: Emergency Room Visit.
RAD: Restricted Activity Days.

# This value has been corrected since implementation; see text, area.
**Table 4.5** Exposure-response functions used for the assessment of chronic morbidity effects due to incremental particulate air pollution.

<table>
<thead>
<tr>
<th>Source</th>
<th>Change in prevalence of</th>
<th>Low</th>
<th>Mid</th>
<th>High</th>
<th>Change in annual PM$_{10}$ concentration per 100000 per year, in [µg/m$^3$]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Schwartz (1993b)</td>
<td>Change in prevalence of adults with chronic bronchitis per 100000 adults, per year</td>
<td>45</td>
<td>70</td>
<td>94</td>
<td>PM$_{10}$</td>
</tr>
<tr>
<td>Dockery et al (1989)</td>
<td>Change in prevalence of children with bronchitis per 100000 children, per year</td>
<td>85</td>
<td>16</td>
<td>23</td>
<td>PM$_{10}$</td>
</tr>
<tr>
<td>Dockery et al (1989)</td>
<td>Change in prevalence of children with chronic cough per 100000 children per year</td>
<td>10</td>
<td>20</td>
<td>31</td>
<td>PM$_{10}$</td>
</tr>
</tbody>
</table>

### 4.9 Comments on Severity/Interpretation of Health Endpoints, in Relation to Economic Valuation

The principal discussion of these issues is given in the methodology report on economic valuation. It may however be worthwhile to re-emphasise that the acute mortality effects occur predominantly in older people, almost certainly with serious pre-existing ill-health conditions, though the precise mechanism of action has not (yet) been established. Length of life lost in those who die prematurely following higher pollution days is also unknown, but is likely to be short relative to average life expectancy. Based on knowledge to date, we speculate that the distribution is strongly skewed to the right, with a median of perhaps several weeks or a few months; much less than the mean, which might well be 12 months or more. This however is conjecture; further research is needed. Average reduced life expectancy among those who die prematurely from chronic effects of air pollution is likely to be much greater, and is estimated somewhat arbitrarily as 12.5 years.

Minor restricted activity days (MRADs: Ostro and Rothschild, 1989) do not involve bed disability, or time off work or school; but in the absence of a specific valuation, is considered equivalent to a RAD. Any symptom, as in Krupnick et al (1990), covers a very wide range of occurrences most of which would not be considered severe. Chronic cough, as in Dockery et al (1989), would be considered less severe than bronchitis; but both are valued similarly. However, that valuation refers rather to an *acute* episode of bronchitis rather than an ongoing condition. The valuation of this and other chronic disease endpoints needs to be re-considered in any future phases of the ExternE Project.
4.10 Aspects of Implementation

There are unquantifiable uncertainties in transferring the results of mostly North American studies to a European context. Some aspects have been considered earlier (4.1.1 above). Others include differences in pollution sources and mixtures, and in the relationships between pollutants and confounding factors. It is however a strength of the studies that relationships, expressed as percentage change in health effect per unit exposure, seem remarkably invariant to changes in population, location and pollution mixtures.

If the exposure-response functions were to be implemented in the form of percentage change in health effects per unit exposure, it would be necessary to know (or to estimate reliably) the background levels of mortality and morbidity over the full population at risk. Death rates are quite readily available; and mortality has therefore been estimated using percentage change functions. It was however impractical to estimate background morbidity rates across the range of endpoints, in a way which took account of possible variations across grid-cells. Instead, a single background rate was estimated from an available epidemiological study (often, but not always, the same study as was used to provide the relationship of percentage change), and combined with the estimated percentage change to give estimated numbers of impacts per unit incremental exposure and unit population.

This approach, while easing implementation, is inexact in that it assumes both that background rates do not vary importantly by gridcell within the population at risk, and that the average background rates can be estimated adequately using studies in other locations. These assumptions ignore population-related variations in true morbidity levels, as well as socio-economic and cultural differences affecting health service usage, restricted activity, and symptoms reporting. Better estimation of background levels of morbidity may be one useful area of improving the present estimates; for example, by using studies in Europe to provide estimates of background rates of occurrence, even where these studies do not provide appropriate exposure-response relationships.

There are also uncertainties in using general air pollution studies to assess the impacts of small pollution increments from a modern power station, which have a different exposure pattern and pollution mixture from ambient air pollution generally; and in using these same relationships to estimate effects at many hundreds of kilometres from source. There is however no rational basis for limiting the range of application to some pre-defined distance from the power plant.

Despite these difficulties, we consider that an informative quantified assessment can be made with current knowledge; though the uncertainties involved should not be disregarded. The scope of uncertainty will surely be reduced as research in air pollution epidemiology continues.
4.11 Discussion

The damage-function approach adopted in the present study is the logical way to proceed with an evaluation of health impacts provided that the state-of-the-art is sufficiently well developed that suitable functions and background data exist. It is nevertheless useful to consider to what extent the implementation of the available exposure-response functions really does express the full health impacts of the burdens and emissions from the fuel cycle.

4.11.1 Health endpoints evaluated

(a) Acute effects: Lung function health impacts from epidemiological studies have been ignored, because at present there are no usable economic valuations of lung function changes. These effects are possibly transient and relatively minor in health terms however; and the inclusion and valuation of symptoms means that some related, relatively minor, effects have been assessed.

Possibly more important is that the present implementation takes no account of hospital admissions for cardiovascular disease which, given cause-specific results for mortality, might be expected also to increase with particulate pollution. Results from Schwartz and Morris (1995), available too late for inclusion in the present phase of ExternE, should help fill this gap.

Air pollution increases the risk of hospital admissions. Arguably, it may also increase the duration of those admissions; an endpoint which, to our knowledge, has not been studied.

A full range of health endpoints for the effects of ozone was implemented. Some endpoints, such as respiratory hospital admissions, are well-established. Results for mortality, although included, are ambiguous however.

(b) Chronic effects: Chronic mortality and morbidity have been included in a preliminary way only; but sufficiently to indicate that they may be (very) important. It is particularly important that the estimated chronic mortality effects of long-term exposure to particles not considered reliable enough for inclusion in summary tables of effects and of their costs; because, if taken at face value, these are the dominant health impact for fossil fuel cycles, and one of the most important pathways in the evaluation as a whole.

4.11.2 Primary and secondary particles

An important recent advance in the present project, relative to earlier drafts, has been the inclusion of the health effects of secondary particles, formed by reactions involving gases emitted during power generation. For long-distance impacts, these may be substantially more important than the effects of particles directly emitted.
4.11.3 Reliability of core E-R functions/representing uncertainty

Where practicable, we have based the present evaluation on studies of appropriate design, and where basic study data (measurement of air pollution, of health outcomes, of possible confounding factors) are reliable, and which use appropriate statistical methods and appropriate adjustment for confounders. The detailed methodological review of the acute effects of particles gives reassurance about the basic soundness of most of the studies used here.

Furthermore, the reliability of individual E-R functions for use within the context in which they were developed is quantified by estimating for each function a ‘mid’, ‘high’ and ‘low’ value; the upper and lower bounds being based on the standard error (SE) of the estimated regression coefficients.

It is very important to note however that uncertainty as expressed via a standard error is only one component of the overall uncertainty within the estimation process carried out within ExternE. Arguably much more important are the soundness of the core risk model assumptions; the simplifications implied in adapting E-R functions for ease of implementation (e.g., linearising); and other aspects of transferability, i.e., of using E-R functions without the context in which they were developed.

Our assessments of the uncertainties associated with these aspects is at present principally qualitative. It is hoped to move towards more quantitative expressions of uncertainty for these aspects also, as well as towards formalising an approach to aggregating uncertainty over endpoints and pollutants.

4.11.4 Modelling the effects of mixtures/ Which index of particles?

(a) Mixtures generally: Following the impact pathway approach, it is difficult to quantify the impact of an air pollution mixture where little is known about the independence or not of the effects of the component parts. The epidemiological evidence suggests however that the acute effects of particles and of ozone may reasonably be considered as additive in many situations, the confounding between these two pollutants being generally less than that between particles, SO₂ and NOₓ.

Here and elsewhere, it is important to remember that the purpose of the risk model is to provide quantitative estimates of the effects which give good approximate answers when applied in a variety of situations. We do not presume to model the detailed possible interactions that may occur within mixtures of pollutants, and between pollution and other factors.

(b) Particles as a mixture: A particularly important aspect of this concerns particles, which really are themselves a mixture rather than a pure substance; and what index or indices of particles may be best for expressing acute or chronic health effects. The present phase of ExternE uses PM_{10} as the index of choice, an approach which is probably the conventional one at this point.
There are however grounds for considering that the important health effects are due to smaller particles within the PM$_{10}$ range, e.g., PM$_{2.5}$; sulphates. It is possible to provide, from epidemiological studies, E-R relationships linking these with various endpoints; though the range of relevant studies is much less than for TSP or PM$_{10}$. Given that most of the particles considered in the present study, whether emitted directly or formed subsequently, are small, then application of these functions based on PM$_{2.5}$ or sulphates might lead to substantial increases in the estimated acute health effects.

4.11.5 Simplifications of the E-R relationships

(a) **Linearisation:** For ease of implementation, several of the key relationships are linearised and annualised, assuming an independence of background levels. The real situation obviously is more complex. For example, the linearised functions are a good approximation to the reported non-linear exposure-response relationships identified in many of the studies, for background conditions of pollution and of disease similar to those which hold in the regions and populations studied. They are however likely to be less appropriate when applied to different sets of conditions. In particular, if the true relationship is non-linear and if there is wide variation in day-to-day pollution increments, then substantial differences can occur in estimated effects, depending on whether the implementation is linearised to an annual average value, as both the US study (ORNL, 1994) and we have done; or whether the more arduous route is taken, of applying a non-linear function to the distribution of daily values, and aggregating the increase in daily impacts.

(b) **Absence of threshold:** Epidemiological studies are relatively uninformative about health impacts at low ambient concentrations of pollution. This is partly because many studies have been conducted in regions (North America, Southern Europe, China, Central and Latin America) where air pollution is generally higher than in much of Europe, especially Northern Europe; partly because it is harder to detect effects at low concentrations (with associated lower day-by-day variability). The use of no threshold for particles, and more generally the application of linearised functions without threshold to very low increments in particulate air pollution far away from source, may have led to an overestimation of effects.

4.11.6 Transferability

There are general problems with the transferability of exposure-response functions from one air pollution context to another, especially when general urban air pollution results are applied in the specific context of emissions from a power station.

(a) **Variability by endpoint:** The transferability of an exposure-response function depends partly on the health endpoint being considered. Transferability is likely to be better to the extent that the endpoint represents a biological rather than a social event. Health indicators like Restricted Activity Days, Hospital Admissions, Emergency Room Visits etc. partly depend on socio-cultural factors which are likely to vary importantly both between Europe and North America, where many of the studies used have been conducted; and within Europe.
(b) Similarity of population studied/ of background morbidity: Many of the studies (e.g. all those with mortality and hospital usage endpoints) are based on the entire populations of particular cities or regions; or have well-defined restrictions (e.g., those aged 65 years or more). Panel studies of individuals are however based on limited numbers of subjects who may be unrepresentative even of the areas where the original study was conducted. For example, the E-R relationships of Krupnick et al (1990), though applied to adults generally, are based mostly on adults aged 30-45 years. Clearly, this would affect the transferability of the overall E-R relation to a wider population.

Possibly the key question here is the transfer between North America and Europe where, as discussed above, the key issue may well be differences in background rates of morbidity and hospital usage, rather than in the effect of pollution expressed as a percentage change relative to background. This is an important issue, methodologically.

(c) Pattern of pollution mixtures and variability in concentrations over time: The issue of disaggregating an air pollution mixture has been considered above. The size of any errors in estimating effects will depend not only on how well this disaggregation has been done; but also, on how similar are the pollution mixtures across which the E-R functions are being transferred. In this regard, we note especially that the mixtures as emitted from power stations vary by fuel type; but are importantly different from the general background mixtures of pollutants on which epidemiological studies are based. In addition, the pattern of short-term (i.e., within-day) variability in incremental exposure from a power station plume will vary according to distance from source; and, when very near to or very far from source, will be importantly different from the pattern of short-term variability in background exposures. The effects of short-term changes in concentrations of ambient pollutants are not known; but these differences contribute to uncertainty about the reliability of the estimation process.

On the other hand, approximate consistency of findings across a wide range of conditions (e.g., PM$_{10}$ and mortality) is clearly a reassurance.

(d) Distance from source: The modelling of dispersion of pollutants has shown that pollution emitted from a high stack may (directly, or indirectly through the chemical reactions which it causes) add some small positive increment to background pollution even at distances of several thousand kilometres from source. In practice, these increments are much too small to detect by direct measurement. But they may affect very large populations at risk leading, if E-R functions apply, to non-ignorable estimated health impacts.

It is difficult to envisage how such small pollution increments might in practice contribute to severe effects such as mortality or hospital admissions, especially as the ‘peakedness’ of incremental exposure will have been smoothed out following transport over such long distances. Nevertheless, the logic of the E-R framework does not exclude some contribution, if the results are transferable. Hence these long-range effects are included.
4.12 Concluding Remarks

These limitations should not however obscure the real achievements. Following review of an extensive and growing epidemiological literature, usable exposure-response relationships for a wide set of acute health effect endpoints have been found and implemented both for particles and for ozone; with some account of chronic effects of particles also. These relationships indirectly reflect an effect of sulphur dioxide and of oxides of nitrogen also, because relationships for particles take account not only of primary emissions, but also of secondary sulphates and nitrates. All relationships have been implemented over wide geographical areas. Despite the uncertainties highlighted above, some of which are likely to be at least partially resolved as further work continues, the work to date has shown that there is here a methodology which can be used and developed to help in quantifying the public health effects of air pollution from power generation.

The basic impact pathway approach already seems workable. Methodologically, the issues are to do with improvement and refinement rather that with the strategy itself.
4.13 References


Ostro B, Sanchez JM, Aranda C, Eskeland GS. Air pollution and mortality: results from a study of Santiago, Chile. [In press].


Public Health Effects


5. RADIOLOGICAL HEALTH IMPACTS

5.1 Boundaries of the Assessment

The incremental impact of an additional power station, is assessed by considering the first-order processes during the routine operation of each stage of the nuclear fuel cycle. The construction and decommissioning are also evaluated for the electricity generation stage. Accidental situations for the reactor and transportation stages are addressed. Although there can be arguments made for the inclusion of second-order processes, this has not been done on a project-wide basis, except in the fuel cycles where significant impacts would be neglected.

The most important choices for the assessment of the nuclear fuel cycle concern the definition of time and space boundaries. Due to the long half-life of some of the radionuclides, low-level doses will exist very far into the future. These low-level exposures can add up to larger numbers when spread across many people and many years (assuming constant conditions). The evaluation uses the following conservative assumptions:

- Lifestyles in the future will result in the same level of external and internal radiation exposure, as exist today;
- A linear response to radiation exposure at very small doses does exist;
- The dose-response function of humans to radiation exposure will remain the same as today;
- The fraction of cancers that result in death remains the same as today.

The validity of this type of modelling has been widely discussed. The uncertainty of the models increases and the level of doses that are estimated fall into the range where there is no clear evidence of resulting radiological health effects. In addition, the very long time scale presents some problems in the direct comparison of the nuclear fuel cycle with the other fuel cycles for which mainly shorter term impacts are considered. In spite of these drawbacks, it was decided that within the project-wide guidelines followed by all fuel cycles, this type of risk assessment methodology was required. The short-term time scale of 1 year includes immediate impacts, such as occupational injuries and accidents. Medium-term considers the time period from 1 to 100 years and long-term from 100 to 100,000 years. The limit of 100,000 years is arbitrary, however the most significant part of the impacts are included.

The assessment of the impacts for a wide range of distances is not as problematic but must also be taken into account. It has been shown that the distance at which the evaluation stops can have a large influence on the final costs. The partitioning of the spatial scale (local, regional, global) is defined at 100 km and 1000 km from the point of release. The results for each category are mutually exclusive.
The impacts estimated for the nuclear fuel cycle are presented in a time and space matrix. This form of presentation of results ensures that all the important impacts are assessed and allows comparison of results in the categories that are appropriate. It also provides a clear presentation of the level of uncertainty associated with the results.

5.2 Uncertainty

The level of uncertainty associated with the results is due to the uncertainties in the models used, their input data and the lack of information available for some pathways. Each part of the methodology contributes to the uncertainty of the final results. For example, in most cases generalised transfer coefficients and assumptions are taken. The estimates of uncertainty presented in the methodology are based on expert judgements and the range of possible input values. As a general rule, the longer the time span and/or the larger the region considered in the model, the larger the uncertainty in the model and the input data.

5.3 Priority Pathways for Environmental Releases

For the radiological impacts to the public and environment, independent evaluations must be done for each radionuclide in each mode of radionuclide release or exposure. Existing radiological dose assessment methodologies have been adapted to fit the basic framework accepted for all the fuel cycles in the ExternE project. Different models are required to evaluate the impact of severe accidents.

Atmospheric, liquid and sub-surface terrestrial releases are treated as separate pathways. Due to the different physical and chemical characteristics of the radionuclides, each nuclide is modelled independently and an independent exposure of dose is calculated. The damage to the general population (collective dose) is calculated based on assumptions for average adult individuals in the population. Differences in age and sex are not taken into account. The methodology allows for the summation of all the doses before application of the dose response coefficients.

If the data are available, the releases to the environment, or the source term, used in an assessment are the average annual releases based on the data from a number of past years for the specific facility. Otherwise more general information is utilised. It is assumed that the annual release occurs at a constant rate and is representative for the 30-year operational lifetime of the facility.

Figure 5.1 illustrates a generalised flow of contaminants in the environment. In all cases the environmental releases with potential public health impacts fall into the three major categories of (1) atmospheric discharge, (2) liquid discharge into a river or the sea, and (3) land based waste disposal. It is not possible to evaluate all the possible environmental pathways, therefore priority is given to the pathways that are the most significant sources of impacts.
Figure 5.1 Impact Pathways for the Release of Radioactivity in the Environment
These priority pathways can be modelled with varying degrees of complexity taking into account the particular radionuclide released, the physico-chemical forms of the release, the site-specific characteristics, and receptor-specific dose and response estimates. With validated models of the transfer of radionuclides in the environment, many nuclide-specific parameters have been determined. Generalised values applicable to European ecosystems have also been developed. In this project, models and parameter values developed in Europe (NRPB, 1994), US (Till and Meyer, 1983) and by international agencies (UNSCEAR, 1993; ICRP, 1974; ICRP, 1991) are the basis for the input parameters used. Site-specific data are used for population, meteorology, agricultural production and water use.

The result of the pathway analysis is an estimate of the amount of radioactivity (Bq) to which the population will be exposed converted to an effective whole body dose (Sv) using factors reported by the National Radiological Protection Board (NRPB, 1991). The method applied does not accurately calculate individual doses or doses to individual organs of the body. It is intended to provide a best estimate of a population dose (man.Sv) and an estimate of the expected health impacts as a result of those doses.

5.4 Impacts of Atmospheric Releases of Radionuclides on Public Health

The most important impact pathways for public health resulting from atmospheric releases are:

- Inhalation and external exposure due to the radionuclides in the air;
- External exposure from ground deposition;
- Ingestion of contaminated food resulting from ground deposition.

5.4.1 Dispersion

Gaussian plume dispersion models are used for modelling the distribution of the atmospheric releases of radionuclides. Wind roses, developed from past measurements of the meteorological conditions at each site, represent the average annual conditions. This methodology is used for both the local and regional assessments. It is recognised that this is not the best method for an accurate analysis for a specific area; however, for the purpose of evaluating the collective dose on a local and regional level, it has been shown to be adequate (Kelly and Jones, 1985). For the global assessment, general box models for the dispersion of H-3, C-14, Kr-85 and I-129 are used (IAEA, 1985).

5.4.2 Exposure

Inhalation doses to the population occur both at the first passage of the "cloud" of radioactive material and, for the extremely long-lived, slow-depositing radionuclides (H-3, C-14, I-129), as they remain in the global air supply circulating the earth. Human exposure is estimated using the reference amount of air that is inhaled by the average adult (the "standard reference man" (ICRP 23) and nuclide-specific dose conversion factors for inhalation exposure in the local and regional areas (NRPB, 1991)).
External exposure results from immersion in the cloud at the time of its passage and exposure to the radionuclides that deposit on the ground. The immediate exposure of the cloud passage is calculated for the local and regional areas. The global doses for exposure to the cloud are calculated for I-129 and Kr-85. For external exposure due to deposition, the exposure begins at the time of deposition but the length of time that must be included in the assessment depends on the rate of decay and rate of migration away from the ground surface. For example, as the radionuclide moves down in the soil column, the exposure of the population decreases due to lower exposure rates at the surface. The time spent out of doors will also affect the calculated dose because buildings act as shields to the exposure and therefore diminish the exposure. This is a case where the conservative assumption that the population spends all the time outside is taken. The assessment of 100,000 years is considered to be a sufficient amount of time to include the major impacts from this pathway.

The human consumption pathway via agricultural products is due to direct deposition on the vegetation and migration of the radionuclides through the roots via the soil. Again, depending on the environmental and physical half-lives of each radionuclide, the time scale of importance varies but it is considered that 100,000 years takes into consideration almost all the possible impacts from long-lived radionuclides.

A detailed environmental pathway model is not used. The environmental transfer factors between deposition and food concentration in different food categories, integrated over different time periods, assuming generalised European agricultural conditions is obtained from the NRPB agricultural pathway model FARMLAND. A constant annual deposition rate is assumed and the variation between different seasons of the year is not taken into account. The agricultural products are grouped, for this generalised methodology, as milk, beef, sheep, green and root vegetables and grains.

Cultivated vegetation is either consumed directly by people or by the animals which ultimately provide milk and meat to the population. The exposures received by the population are calculated taking into account food preparation techniques and delay time between harvest and consumption to account from some loss of radioactivity. An average food consumption rate and population size are used for calculating the amount of food that is consumed in the local, regional and global population. The collective doses are calculated assuming that the food will be consumed locally but if there is an excess it will pass to the regional population next, and afterwards, to the global population group. In this way the dose due to the total food supply produced within the 1000 km area included in the atmospheric dispersion assessment is taken into account, even if the exact location of consumption is unknown.

5.4.3 Dose assessment

It is possible to report a calculated dose by radionuclide, type of exposure and organ of the body, but for the purpose of this project, an average individual whole body effective dose is used to calculate the collective (population) dose.

The relationship between the dose received and the expected number of radiological health impacts is based on the information included in the international recommendations of the ICRP 60 (ICRP, 1991). The factors, or dose response functions, used to predict the expected
occurrence of cancer over a lifetime or severe hereditary effects in future generations per unit exposure received by the general public are 0.05 fatal cancers per man.Sv (unit of collective dose) and 0.01 severe hereditary effects in future generations per man.Sv. These factors assume a linear dose response function.

The number of cancers that would be expected to be non-fatal (0.12 non-fatal cancers per man.Sv) are calculated based on the expected number of fatal cancers and the lethality fractions reported for 9 categories of cancer in ICRP 60. This is reflected in the aggregated non-fatal cancer factor of 0.12 per man.Sv.

It is recognised that the dose-response functions that are chosen in the assessment of radiological health effects are extremely important. There is still controversy on the exact values to use and different models have been proposed. Within the context of this project, the internationally accepted factors are used. Further details are presented in the nuclear fuel cycle report (European Commission, 1995).

5.4.4 Time distribution of the expected occurrence of health effects

The use of the dose response functions provides the estimate of the total number of health effects expected, however the details on the expected time of occurrence of these effects are not addressed. The deterministic health effects that occur after high doses of radiation (accidental releases) will occur in the short-term, but the distribution in time of the stochastic health effects is dependant on two factors: (1) the continued existence of radionuclides in the environment for years after deposition, and (2) the latency between exposure and occurrence of the effect.

The distribution of the total number of cancers is statistically predicted over the 100 years after 1 year of exposure, using data for the expected distributions of cancer in the average population as a result of low-level radiation exposure (for example, Figure 5.2 shows the distribution for France).

This curve is integrated over the 30-year operational lifetime of the facilities. After the shutdown of the facilities, except in the disposal stages, the releases do not continue and the level of radioactivity due to the releases decreases in a manner dependant on their physical and environmental half-times.

For practical purposes in this assessment, the decreasing level of exposure is not integrated continuously through time but assumed to be constant in blocks of time of 0-1, 2-30, 30-50, 50-100, 100-200 and 200-100,000 years after the operational releases. The final accounting of potential cancers ranges from the first year of release to 300 years into the future.
Figure 5.2 Relative Frequency of Occurrence of a Cancer for a French Population after 1 Year Exposure.

This methodology may slightly underestimate the economic value of cancers due to the assumption of a constant exposure rate during the block time periods. In reality, the exposure will be greater at an earlier time in the block than at the end; however, the break down of time periods has been chosen to minimise the difference. Estimates of the occurrence of severe hereditary effects during the next 12 generations are made using information presented in ICRP 60 (for further details, see European Commission 1995).

5.5 Impacts of Liquid Releases of Radionuclides on Public Health

Depending on the site of the facility, liquid waste is released into a river or the sea. The priority pathways for aquatic releases are the use of the water for drinking and irrigation, and the consumption of fish and other marine food products.

For the marine environment, the seafood and fish harvested for human consumption is the only priority pathway considered in this assessment. The other possible pathways involving the recreational use of the water and beaches do not contribute significantly to the population dose and are not considered as a priority.

5.5.1 River

The dispersion of a release to a river is modelled using a simple box model that assumes instantaneous mixing in each of the general sections of the river that have been defined. The upstream section becomes the source for the downstream section. River-specific characteristics, such as flow rate of water and sediments, transfer factors for water/sediments and water/fish, are needed for each section. The human use factors such as irrigation, water treatment and consumption, and fish consumption must also be taken into consideration.
The deposition of radionuclides from irrigation water onto the surface of the soil and their transfer to agricultural produce is assumed to be the same as for atmospheric deposition.

The ingestion pathway doses and health effects are estimated in the same way as described in the previous section on the atmospheric pathway. For aquatic releases, it is difficult to calculate independent local and regional collective doses without creating extremely simplified and probably incorrect food distribution scenarios. Therefore, the local and regional collective doses are reported in the regional category.

### 5.5.2 Sea

To evaluate the collective dose due to consumption of seafood and marine fish, a compartment model which divides the northern European waters into 34 sections is used. This model takes into account volume interchanges between compartments, sedimentation, and the radionuclide transfer factors between the water, sediments, fish, molluscs, crustaceans, and algae, and the weight of fish, molluscs, crustaceans and algae harvested for consumption from each compartment. For the regional collective dose, it is assumed that the edible portion of the food harvested in the northern European waters is consumed by the European population before any surplus is exported globally. Due to the difficulty in making assumptions for the local consumption, the local collective dose is included in the regional results. Marine dispersion at the global scale of H-3 and I-129 is evaluated using general global dispersion models (IAEA, 1985). The risk estimates of this pathway are calculated using the same methodology as described for the other pathways.

### 5.6 Public Health Impacts of Releases of Radionuclides from Radioactive Waste Disposal Sites

The land-based facilities designed for the disposal of radioactive waste, whether for low level waste or high level waste, are designed to provide multiple barriers of containment for a time period considered reasonable relative to the half-life of the waste. It is assumed that with the normal evolution of the site with time, the main exposure pathway for the general public will be the use of contaminated ground water for drinking or irrigation of agricultural products.

The leakage rate and geologic transport of the waste must be modelled for the specific facility and the specific site. The global doses due to the total release of H-3, C-14 and I-129 are estimated assuming that ultimately the total inventory of wastes are released into the subsurface environment in liquid form and using the same global dispersion models as used for the liquid releases of H-3 and I-129 to the ocean. As with the other pathways, it is assumed that the local population and their habits remain the same for the 100,000-year time period under consideration for the disposal sites. This time limit takes into account disposal of all the radionuclides except long-lived I-129.
5.7 Impacts of Accidental Atmospheric Releases of Radionuclides

Accidental releases are evaluated using a risk-based expected damages methodology. Risk is defined as the summation of the probability of the occurrence of a scenario \( (P_i) \) leading to an accident multiplied by the consequences resulting from that accident \( (C_i) \) over all possible scenarios. This can be simply represented by the following equation:

\[
\text{Risk} = \sum P_i \cdot C_i
\]

5.7.1 Transportation accidents

In the analysis of transportation accidents, a simple probabilistic assessment is carried out. It is not possible to evaluate all possible scenarios for the accident assessments but a representative range of scenarios, including worst case accident scenarios, need to be included. The type of material transported, the distance and route taken by the train or truck, the probability of the accident given the type of transportation, the probability of breach of containment given the container type, the probability of the different type of releases (resulting in different source terms) and the different possible weather conditions are taken into account. The site of the accident can play a key role in the local impacts that result so variation in the population and their geographic distribution along the transportation routes is considered.

The atmospheric dispersion of the release is modelled using a Gaussian plume puff model. The toxicological effects of the releases (specifically UF\(_6\)) are estimated using the LD\(_{50}\) (lethal dose for 50% of the exposed population) to estimate the number of expected deaths and a dose-response function for injuries due to the chemical exposure. The radiological impacts are estimated using the same methodology described previously for an atmospheric impact pathway. The expected number of non-radiological impacts, such as death and physical injury due to the impact of the accident is estimated. These types of impacts are also calculated for the transportation of non-radioactive materials (concrete, steel) involved in the construction and decommissioning of the reactor.

The general methodology for the assessment of transportation risks is presented in Figure 5.3. The single cost value presented takes into account transportation between all stages of the fuel cycle. The cost of clean-up and countermeasures that might have to be taken are not included.

5.7.2 Severe reactor accidents

A comprehensive probabilistic safety assessment (PSA) of potential reactor accidents does not fall within the scope of this project. In addition, the detailed data on potential source terms and associated probabilities for a multitude of potential scenarios for European nuclear power plants are not available. As result, four hypothetical scenarios have been evaluated in order to demonstrate the range of results obtained using a risk-based assessment methodology. The scenarios were assumed to take place at a hypothetical power plant in the centre of western Europe.
Figure 5.3 Pathway for the Assessment of a Transportation Accident.
The probability assumed for an accident resulting in a core melt at a nuclear reactor has a major influence on the results of the assessment. The more modern 1300 MWe reactors are considered to have a lower probability of occurrence for a core melt accident than the older 900 MWe models; in this study a probability of 1E-5 per reactor-year was used (EDF, 1990). This is smaller than the estimated value of the NRC (NRC, 1990) but significantly higher than most of the probability values considered to be correct for a present-day European reactor (Wheeler and Hewison, 1994).

The magnitude and characteristics of radioactive material that can be released following a core melt will depend, inter alia, on the performance of the containment and its related safety systems. If the containment suffers massive failure or is bypassed, a substantial fraction of the volatile content of the core may be released to the environment, if the containment remains intact the release will be very small. For the purposes of this indicative assessment, it was assumed that the probability of massive containment failure or bypass conditional upon a core melt was 0.19, and the probability of the containment remaining intact was 0.81 (NRC, 1990). The same assumptions were made for the 900 MWe PWR assessment (European Commission, 1995a).

Massive containment failure was assumed to result in the total release of noble gases from the core, 10% of the more volatile elements, such as caesium and iodine, and smaller percentages of other elements. The other characteristics of the release (e.g. height, duration, release fractions of non-volatile, etc.) were taken, for convenience, to be those assumed for the source term ST2 in a recent CEC/NEA inter-comparison study (OECD, 1994). Releases a factor of 10 and 100 times lower were also assumed in order to evaluate the sensitivity of the results to the assumed release fractions. For an intact containment 0.1% of the noble gases and 0.01% of the more volatile elements were assumed to be released.

The public health impacts and economic consequences of the releases were estimated using the European accident consequence software COSYMA (Ehrhardt and Jones, 1991). One hundred and forty-four different meteorological scenarios were statistically sampled to predict the dispersion of the releases. Due to the introduction of countermeasures for the protection of the public the impact pathway needed to be altered, as shown in Figure 5.4. The priority atmospheric release pathways, for local and regional areas out to 3,000 km from the site were assessed. Unfortunately the definition of time and space boundaries are not the same as those defined in the assessment of routine operations of the fuel cycle, so the results need to be presented separately.

The monetary valuation of the health effects arising from the collective dose was completed in the same manner used throughout the assessment. The additional costs from the implementation of the countermeasures and the agricultural losses were calculated by COSYMA using estimates of the market costs. A full description of this assessment and the results obtained are presented in the nuclear fuel cycle report (European Commission, 1995).
Figure 5.4 Pathways for a Severe Accidental Release.
The use of this type of methodology does not necessarily include all the social costs that might result after a severe nuclear accident. One important issue is the social costs of risk aversion. Further work is required before the external costs of a severe accident can be considered complete.

5.8 Occupational Impacts

The assessment of the radiological and non-radiological occupational impacts are very straightforward because modelling is not required for this pathway. The radiation protection of the workers requires direct monitoring and reporting of the doses received by the workers at each facility thereby providing measured data for the evaluation of radiological impacts. The only exception is for waste disposal where the estimates of occupational doses are taken from a UK study (Ball et al, 1994).

The relationship between the dose received by the occupational population and the radiological health impacts are based on information published in ICRP 60 (ICRP, 1991). The factors, or dose response functions, used to predict the expected occurrence of cancer over a lifetime or severe hereditary effects in future generations per unit exposure received by the workers are 0.04 fatal cancers per man.Sv and 0.006 severe hereditary effects in future generations per man.Sv.

The fraction of cancers that would be expected to be non-fatal are calculated based on the expected number of fatal cancers and the lethality fractions in the worker population reported for 9 categories of cancer reported in ICRP 60. The different age and sex distributions found in the working population compared to the general public slightly change the expected occurrence of disease of all three types of health impacts. This is not easily seen in the aggregated non-fatal cancer factor of 0.12 per man.Sv because the public and worker values are rounded.

The methodology for estimating the latency time before occurrence of the health effects has been presented earlier.

Non-radiological worker accidents data are obtained from the specific facility. If insufficient data are available for a representative value, the national accidents statistics (for example, in France CNAM, 1991) reported by type of job can be utilised. When it is not possible to find the data for a specific nuclear facility, the data for the chemical industry can be used as an approximation.

For the construction and the decommissioning of the reactor, the workforce is calculated based on the construction and decommissioning costs associated with the worker productivity in each industrial branch involved in the works, and normalised by the electricity production of the plant over its lifetime (30 years).
5.9 Impacts of Transportation on Human Health

The priority impact pathway from accident-free transportation operations in the nuclear fuel cycle is external exposure from the vehicle containing the radioactive material. This is estimated using a computer code from the International Atomic Energy Agency (INTERTRAN) which takes into account the content of the material transported, the type of container, mode of transport (road or rail), the distance travelled, and the number of vehicle stops at public rest stations along the highway (for road transportation).

5.10 Monetary Valuation

The results of the human health impact analysis are reported as deaths, fatal cancers, non-fatal cancers, severe hereditary effects, occupational accidental deaths and injuries (working-days-lost and permanent disabilities). The value of a statistical life (VSL = 2.6 MECU) is used for deaths, accidental deaths and fatal cancers. This is the same value that is applied in all the fuel cycles being assessed in the ExternE project (see Part II of this report).

Being that no values for the willingness-to-pay to avoid non-fatal cancers have been identified for Europe, the values reported in the US Nuclear Fuel Cycle report (ORNL, 1993) for direct (hospital, physician, drugs, etc.) and indirect (fought earnings discounted at 6%) costs of different types of cancers are used. Since the total average costs per cancer differ by about a factor of 2, it is considered adequate to take the rounded value of a simple average of the total costs as the value of a non-fatal cancer, 0.25 MECU.

For occupational injuries, values per working-day-lost and per permanent disability can be derived from published statistics on compensation for injuries and insurance payments. (In France values have been estimated as 65 ECU per working-day-lost and 19,000 ECU per permanent disability, (European Commission, 1995)). Injuries estimated from traffic accidents occurring during the transportation of material between stages are valued based on statistics from insurance company payments. For example, in France these have been valued at 15,000 ECU. Further work is needed to determine if the use of these values provides the external costs due to these physical impacts. The US team used a "revealed preference" analysis using hedonic wage techniques (ORNL, 1993). An industry average for the electricity generation sector of a value of a statistical injury (VSI) of 10,300 US$ (range of 8,000 - 34,000 US$) was used. It is assumed that in the mining industry, where the activities are more dangerous, the best estimate of a VSI would be 21,000 US$. The values used in the EC assessment can be generally considered to be the same within an order of magnitude. The discounting of these health effects costs are based on the expected distribution of occurrence through time.

In the valuation for the nuclear fuel cycle, it is assumed that the severity of a hereditary effect merits the same valuation as the VSL (2.6 MECU). The discounting of severe hereditary effect costs is much more complex because none of the impacts are seen during the lifetime of the population that is exposed. Data show that 15% of the cases that may occur are expected
to be seen during the first generation, 12% during the second generation and the remaining 73% sometime in the future (ICRP, 1991). For the purpose of applying the 3 and 10% discount rates, the 73% remaining impacts are assumed to occur during the next 10 generations at a constant rate (7.3% per generation out to a total of 12 generations after exposure). The modified VSL-severe hereditary effect values for the 3% and 10% discount rates are 0.296 MECU and 0.039 MECU, respectively. Again full details of the values used are discussed in the nuclear fuel cycle report (European Commission, 1995).
5.11 References


Electricite de France (1990), Etude probabiliste de sûreté des REP de 1300 MWe, EPS 1300, France, May 1990.


6. OCCUPATIONAL HEALTH EFFECTS

6.1 The Impact Pathway

Occupational health effects are caused by accidents and by occupational diseases. Occupational accidents involve an immediate direct physical impact on the worker, with an obvious relationship between cause and effect. In contrast, occupational diseases generally occur as a more or less delayed response to a continuous, often long-term, exposure of an external burden, e.g. airborne pollutants, noise, vibration, etc. Because many of the occupational diseases are typical for a certain activity and normally the affected person is mainly exposed at work, a quantification of impacts is sometimes possible based on statistical data provided by the occupational health insurance system or health and safety reporting system. However, as far as reliable dose-response functions and the relevant input data are available, impacts should be estimated, where possible, following the damage function approach used elsewhere in this study.

The calculation of occupational health impacts based on statistics is regarded as a special case of an impact pathway, in which health data are linked to activities required for electricity production. Calculating occupational health impacts from statistical data implicitly assumes that the marginal impact of an additional facility is equal to the average impact. Provided that data is chosen carefully to be representative of current practice, this assumption should be reasonably valid.

The following factors need to be considered when deciding on the time frame over which data should be collected:

- Changes in technology;
- Changes in legislation, guidelines or working practices;
- Changes in statistical reporting practices or procedures;
- Number of cases (so as to ensure large enough data sets);
- Variation between years.

For example, the Piper Alpha disaster led to significant changes in UK offshore working practices; thus data from the years before this accident will not be representative of current safety levels in the industry. This also highlights the problem of variation between years - average fatality statistics in the sector will be significantly skewed by the accident data from this year. There is, however, some potential conflict between the above list of factors. To provide a statistically reliable sample, data should ideally be taken from as long a period as possible, but such a period will be interrupted by changes in legislation and technology which may cause significant and abrupt changes in accident rates.

In all cases, the level of analysis for different activities or fuel cycles for occupational health effects depends on the available statistics. At an early stage of the ExternE Project it became clear that there are considerable differences between the classification systems used in different
countries. Cultural differences may also lead to variation in the extent to which accidents are reported. For this reason, an understanding is needed of the basis for collecting and reporting statistics in each country to enable a fair and valid comparison to be made. Data requirements for assessment of occupational health impacts from statistical data are shown in Table 6.1.

Table 6.1 Reference data requirements for the estimation of occupational health impacts, using the example of the coal fuel cycle.

<table>
<thead>
<tr>
<th>Activity</th>
<th>Reference data required</th>
<th>Output units</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mining</td>
<td>1. Material and services required for electricity generation.</td>
<td>1. Tonnes/TWh, Tonne kms/TWh, Man years/TWh, etc.</td>
</tr>
<tr>
<td>Transport</td>
<td>2. Productivity data.</td>
<td>2. Tonnes/year, Tonne kms/year, Man years/TWh, etc.</td>
</tr>
<tr>
<td>Electricity</td>
<td></td>
<td></td>
</tr>
<tr>
<td>production</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Waste disposal</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Construction</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Dismantling</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Endpoints</strong></td>
<td>Data on occupational accidents and diseases.</td>
<td>Occupational accidents/year</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Occupational disease cases/year</td>
</tr>
</tbody>
</table>

A further problem for the ExternE Project team concerns identification of accident and disease rates for countries outside of the European Union, such as Australia, Colombia, Poland, South Africa, Saudi Arabia and the USA. All of these countries export fuel to the EU. Occupational health records in these countries range from being comparable to EU standards to being significantly worse. It is therefore necessary to quantify impacts associated with occupational health issues specifically for all countries active in a given fuel cycle; the use of a supposedly generic figure is likely to provide results that are highly misleading. This raises difficulties for data collection, but these must be overcome if the study is to provide reliable information.

There is also an additional complication when analysing the ‘external costs’ of occupational effects, because of the possibility that occupational health impacts are internalised by the action of labour or insurance markets. This is an important issue, but is beyond the scope of the current study.

For some types of occupational disease a more detailed assessment is possible using the impact pathway approach. This has been possible for underground coal mining for the simple reason that this industry has several centuries of operational experience to draw on. It also has well documented and distinct long-term health risks. These long-term risks cannot be accurately quantified from current occupational disease rates, as the health endpoints were induced by historical levels of occupational exposure (i.e. such as dust levels) rather than from current levels in modern mines. Table 6.2 provides details of the information needed for a detailed assessment of such cases.

However, with the exception of coal mining, a detailed damage-function approach to occupational disease has not so far been possible in the fossil and renewable fuel cycles. The main reason for this is a lack of readily available data. For some fuel cycles, this is thought to
significantly underestimate potential impacts; for example, there are certainly long-term occupational diseases which occur in professional divers in the offshore oil and gas industry. It is hoped that the extension of the methodology to such endpoints using a damage function-approach will be possible in the future.

Table 6.2 Reference data and model requirements for the estimation of health impacts in coalminers from long term exposure.

<table>
<thead>
<tr>
<th>Activity</th>
<th>Reference data required</th>
<th>Type of models or relationships</th>
<th>Output units</th>
</tr>
</thead>
<tbody>
<tr>
<td>Under ground coal mining</td>
<td>Total concentration of relevant pollutant</td>
<td>Measured values</td>
<td>[mg/m³]</td>
</tr>
<tr>
<td>Exposure</td>
<td>Exposure time</td>
<td>Sum of exposure concentrations over time</td>
<td>[gh/m³]</td>
</tr>
<tr>
<td>Impact assessment</td>
<td>Population at risk</td>
<td>Dose-response functions</td>
<td>Cases/year</td>
</tr>
</tbody>
</table>

The following section describes the approach used for assessing health effects from coal mining. It provides an example of the detailed use of the impact pathway approach for categorising occupational disease.

6.2 Long Term Health Effects of Coalworkers

6.2.1 Introduction

This section discusses the available exposure-response relationships linking health impacts in coalminers, with long-term occupational exposures. It does not consider the effects of accidents, which are treated separately later in this Chapter. The main occupational health effects from long-term exposure in the coal industry can be divided into:

- Mortality:
  - from exposure to radon (lung cancer); from other occupational exposures (lung cancer); and
  - from other non-malignant, non-violent causes.

- Morbidity:
  - including simple pneumoconiosis (CWSP) and its advanced or complicated form progressive massive fibrosis (PMF); respiratory symptoms; and other occupational diseases.

The evaluation carried out in the companion study for the US DOE (ORNL/RFF, 1994) estimated the lung cancer risks associated with coalminers’ occupational exposures to radon. They considered the health effects of coalminers’ exposures to dust, but did not proceed to assessment of impacts because of difficulties in predicting the incidence of disease attributable to an incremental demand for coal, for example from extrapolating data on coalminers compensated for lung disease which occurred when dust exposures were substantially higher, to future conditions at low exposures. The difficulties of extrapolation identified by the US team (e.g. who qualifies as a coalminer? what severity of disease leads to compensation? is disease
attributable solely to coalmine work? how do time-lags affect the predictions?) are real problems. These problems can, however, be overcome if relationships exist that reliably link the occupational exposures of coalminers with pneumoconiosis and other diseases.

Such relationships do exist for miners in underground coalmines, based on research in several countries. Given current European practice, underground coal mining is relevant to the case studies that have been considered so far by the ExternE Project. However, care needs to be taken in applying this data to predict occupational disease rates in open-cast mining, as it is not clear to what extent the relationships will be transferable to surface coal mining contexts. Given the very different patterns of exposure of miners underground and on the surface, it is likely they will be significant.

From the available research we have, where possible, used the key relevant relationships derived from the UK long-term Pneumoconiosis Field Research (PFR) programme (e.g. Jacobsen, 1981). This is not simply because of the obvious relevance of the PFR to conditions in England. The scope and scale of the PFR, and the reliability of its dust exposure data, give the programme a particularly important place in occupational epidemiology. For example, the dust standards in US underground coalmines are based on PFR results and recent US papers give results similar to those of the PFR.

The available relationships linking dust exposure and morbidity necessarily take account of smoking habit and other factors. No attempt is made to incorporate risk avoidance behavior at the individual level, e.g. the wearing of respirators during work underground. As a result, risks may be somewhat over estimated.

**6.2.2 Coalminers’ mortality**

*Exposure to radon and lung cancer*

The US evaluation considered the relationship between coalminers’ exposures to radon and mortality from lung cancer. The risk estimates used by the US team to estimate lung cancer risks of miners’ occupational exposures to radon were those of the Time Since Exposure (TSE) model of the US Committee on the Biological Effects of Ionizing Radiation (NRC, 1988), known as BEIR IV. Samet (1991) reviews BEIR IV and five other risk estimation models for radon exposure and lung cancer, with special emphasis on NCRP (1984) and ICRP (1987) as the main alternative models; BEIR IV, NCRP and ICRP are also reviewed by Harley (1992). Of these three, the ICRP model gives the highest risk estimates but it is also the least complex, to the point of biological implausibility. Both NCRP and BEIR IV models take account of time since exposure, the NCRP model giving lower risk estimates. Harley (1992) considers that BEIR IV over-estimates the lifetime risks by a factor of about 1.7 because of an error during its derivation.

BEIR IV and the other current risk models for radon and lung cancer are derived principally from studies of miners of various ores: BEIR IV is based on four studies of uranium and iron-ore miners. Studies of fluor spar and tin miners are also relevant. Application of these models to the coal mining situation requires extrapolation from high to low exposure, because the radon exposures of coalminers, estimated by O’Riordan (1991) as about 0.2 Working Level Months
Occupational Health Effects

(WLM) per year worldwide, are very substantially lower than those of the uranium, iron and other ore miners studied. The alternative of using risk estimates from epidemiological studies of domestic exposure to radon, where concentrations are more similar to or lower than those experienced occupationally by coalminers, is not yet feasible. Results from such studies have been published (Axelson et al, 1979; Edling et al, 1984) but the risk estimates are not reliable, mainly because of the difficulty of estimating domestic exposure reliably.

Maclaren (1992) directly studied the lung cancer mortality of coalminers in relation to estimates of their occupational exposures to radon and thoron. Analyses of lung cancer were based on 521 deaths in a cohort of 12,361 coalminers who attended medical surveys at any of 10 coalmines in Britain in the 1950s and early 1960s. The analyses considered the effects of individuals’ exposures to low levels of radon and thoron, estimated from detailed work histories but only limited measurements of radon and thoron daughters, and adjusting for age and smoking habits. Different approaches to the analysis did not show any consistent increase in lung cancer risks overall in relation to radon or thoron exposure, though risks as estimated by the BEIR IV model fell within the 95% confidence intervals of the Maclaren study. Case-reference analyses did show slightly elevated risks with increased radon exposure in non-smokers and light smokers, but not in heavy smokers.

Thus, if risk estimates are to be used, they will necessarily be based on existing models for exposure to radon and lung cancer. Given the need for extrapolation from high to low exposures, biological plausibility is an important criterion. Consistency with the US approach is also desirable. Consequently we recommend use of the BEIR IV model. Maclaren’s results give some support to Harley’s view that BEIR IV may overestimate effects; but they also suggest that application of BEIR IV to the coal mining context gives results of the right order of magnitude.

The US evaluation reproduces the Tables from BEIR IV which are necessary for risk estimation. These Tables have been derived using data on smoking habits, and death rates, for the US population 1980-84. We do not believe that the use of US population data for 1980-84 is a serious obstacle to using the BEIR IV risk estimates in the European context. What is needed is a reference population whose experience is similar enough to the reference populations for the specific reference regions in terms of age structure, smoking habit and mortality. That similarity is not primarily to the reference populations as they are at present, but rather as they will be over the next 40 years; and on this basis, the 1980-84 US experience may be as applicable to Europe as it is to the USA itself. Furthermore, the BEIR IV model estimates relative risks; and small changes in background mortality are unlikely to make an important difference to the incremental results. Changes in smoking habit, which in turn affect lung cancer and other mortality, are likely to be the more serious. Finally, there are grounds of expediency: it is not cost-effective within the present project to develop specific results using UK or German smoking and mortality data.

For this reason, we use the BEIR IV tables of risk estimates as produced in the US draft report. These were presented in the Coal Fuel Cycle Report (European Commission, 1995a).

Coalminers’ mortality from cancer and occupational exposures: other impact pathways.
There is no evidence that coalminers’ mortality from lung cancer is related to their exposures to respirable coalmine dust (Miller and Jacobsen, 1985). Coalminers’ may nevertheless be exposed occupationally to other known or suspected lung carcinogens, notably quartz and diesel fumes. However, there are as yet no suitable epidemiological studies of coalminers’ mortality from which risk estimates linking lung cancer with exposure to quartz or to diesel fumes might be derived. Nor, at present, are there well-accepted exposure-response relationships from studies of workers in other industries, which could be applied to the experience of coalminers in Europe. For example, the major US mortality study of railroad workers (Garshick et al, 1988) shows a clear trend of increasing lung cancer relative risks with the railroad workers’ years of exposure to diesel exhaust fumes. Mauderley (1992) attempts to derive general exposure-response models from this and other studies. However, it is not clear how these results transfer to the coal mining context, and other studies of workers exposed to diesel fumes have produced less convincing results. The situation should be kept under review.

Studies of coalminers in several countries have pointed to an association between coal mining and cancers of the digestive organs (e.g. Jacobsen, 1976, and Miller and Jacobsen, 1985, in the UK; Matolo et al, 1972, and Rockette, 1977 in the USA; and Meijers et al, 1991, in the Netherlands). Again, none of these studies can be used straightforwardly for risk estimation purposes. We know of no established findings for mortality from cancers of other sites, though an association between diesel fumes and bladder cancer is possible (Mauderley (1992).

Coalminers’ mortality from non-malignant causes, or from non-violent causes generally

With regard to mortality generally, Miller and Jacobsen (1985) showed that the relative risks of mortality from all non-violent causes was increased for men with progressive massive fibrosis (PMF) compared with men without simple pneumoconiosis (i.e. CWSP Cat 0). The relative risks were age-dependent. For men with PMF ‘A’ shadows aged 25-34, 35-44, 45-54 and 55-64 respectively at the start of the study, the risks of mortality over the subsequent 22 years were estimated as about 3.5, 1.7, 1.2 and 1.2 relative to men with CWSP 0 from corresponding age-groups. The excess mortality was attributable to causes other than lung cancer or ischaemic heart disease.

The mortality rates for men with more advanced PMF (i.e. ‘B’ or ‘C’ shadows) at start of follow-up were higher than for those with PMF ‘A’ shadows at that time. Death rates for men with CWSP Cat 3 were also higher than for those with Cat 0 initially; but the evidence of increased mortality in men with CWSP 2 initially was not clear.

Miller and Jacobsen also showed increasing risks of mortality from non-violent causes generally, and specifically from non-malignant lung disease (pneumoconiosis; bronchitis and emphysema), with increasing exposures to respirable coalmine dust accumulated up to the start of the study follow-up period. There is a need for care in using their results for risk estimation because, though relationships have been demonstrated clearly, the exposures were not updated to take account of exposures experienced after the study began. This is likely to have least affected those men aged 55-64 years at the start of follow-up, who by that time had accumulated the greater part of their lifetime’s exposure to respirable dust. However, the estimation of relative risks for this subgroup is complicated by the small numbers at risk in the lowest dust exposure
ranges, and so we have not attempted to derive an exposure-response function for implementation in the present project.

Further analyses of existing data might well provide usable and useful exposure-response functions linking mortality from non-malignant causes with exposure to dust and/or radiological signs of lung disease. For the purposes of the present project, we simply draw attention to the findings on CWSP, PMF and mortality as background to the economic evaluation of respiratory morbidity in coalminers.

6.2.3 Coalminers’ morbidity

Coalminers’ exposure to dust is related to several different epidemiological measures of lung disease. First, it can lead to coalworkers’ simple pneumoconiosis (CWSP) and to its advanced or complicated form, Progressive Massive Fibrosis (PMF), both identified from X-ray films of the lung; i.e. the ‘black lung’ discussed in the US report. Secondly, it can lead to chronic bronchitis as determined from the presence of respiratory symptoms and to the respiratory symptoms of breathlessness. Thirdly, exposure to respirable coalmine dust is associated with loss of lung function.

The relationship between these conditions is complex, but clearly more than one disease process is involved. For example, exposure to respirable coalmine dust is associated with loss of lung function even in miners without simple pneumoconiosis (Soutar and Hurley, 1986); and in miners generally, there is no residual loss of lung function associated with CWSP (understood as the presence of small rounded opacities) once adjustment has been made for the higher than average dust exposures of miners with CWSP (Rogan et al., 1973; Collins et al., 1988). PMF is however a severely disabling disease, and is associated with lung function losses over and above those attributed independently to the dust exposure which gave rise to PMF (see e.g. Hurley, 1985).

The relationship between respiratory symptoms and CWSP was investigated by Collins et al. (1988). Having adjusted for age, smoking habit and dust exposure, they found that the relative risk that a miner would report respiratory symptoms of breathlessness was 4.1 in miners with the small rounded opacities typical of CWSP, compared with miners with no small opacities. The corresponding relative risk for the symptoms of chronic cough with phlegm, but without breathlessness, was 2.2. Thus, CWSP is associated with respiratory symptoms.

With regard to lung function and symptoms, statistically significant losses of lung function associated with dust exposure have been found even in miners without the respiratory symptoms of chronic bronchitis (Soutar and Hurley, 1986; Marine et al., 1988). Furthermore, in the 3,380 coalface workers studied by Marine et al., the overlap between men who reported symptoms of chronic bronchitis and those whose lung function was 80% of predicted was about 50% of either group; and only about 33% in non-smokers. Thus, many miners with lung function losses do not report symptoms of chronic bronchitis.

For the implementations to date, estimates of risks of developing CWSP or PMF over a working life, have been calculated using results from Hurley and Maclaren (1987). We also consider the dust-related risks of respiratory symptoms, first of breathlessness, then of chronic cough and
phlegm but without breathlessness, in miners without CWSP or PMF, using results from Collins et al (1988). These various impacts are by definition additive from the viewpoint of economic valuation. Estimates are not given for the dust-related increase in respiratory symptoms in miners with CWSP or PMF; if necessary, these impacts can be considered in the economic valuation of CWSP and of PMF. We also ignore dust-related lung function losses, because at present these cannot be used directly in economic valuation. It is clear however that some dust-related lung disease is thereby ignored. There is no body of evidence indicating that asthma is an occupational disease of coalminers.

**Coalworkers’ simple pneumoconiosis (CWSP) and progressive massive fibrosis (PMF): exposure-response relationships**

CWSP of increasing severity is classified as Category 0, 1, 2 or 3 according to criteria developed by the International Labour Office (ILO). Category 0 may include early signs of simple pneumoconiosis, but in practice it may be interpreted as absence of economically relevant disease. A more refined classification, based on the now standard elaborated (12 point) ILO scale, has not yet been used for the exposure-response relationships presented.

CWSP is not necessarily a disabling condition; a miner may well be unaware of its presence unless a suitable chest X-ray film is taken and assessed. It is however associated with respiratory symptoms, and the risks of developing PMF increase with attained category of CWSP. PMF is disabling and, as noted above, leads to reduced life expectancy.

It would seem that CWSP is nevertheless relevant to the present project. Miners or ex-miners with sufficiently severe forms of the disease qualify for compensation in the industrialised western countries with coal mining industries. In the UK, compensation begins when Category 2 or higher CWSP is identified, irrespective of lung function losses or presence of respiratory symptoms. There may in addition be a willingness to pay to avoid these, and earlier, signs of the disease.

Hurley and Maclaren (1987) estimated the risks of developing CWSP or PMF in the course of a 40 year working life as a coalminer exposed to various dust concentrations. Their source data were 52,264 five-year study periods generated by more than 30,000 miners employed at 24 British coalmines for periods between 1953 and 1978; and who in that time had attended two or more successive five-yearly radiological surveys as part of the PFR research programme. Dust exposures were estimated from detailed individual work history records linked with occupational and time-specific dust concentrations from long-term and intensive dust monitoring at the 24 collieries. The X-ray films were classified as Categories 0, 1, 2 or 3 CWSP, or PMF, according to the ILO classification scheme.

Hurley and Maclaren used a Markov-type approach to estimate the dust-specific risks of developing CWSP or PMF over a working lifetime. They first used logistic regression methods to estimate the probabilities of radiological changes over five-year periods in relation to age and cumulative exposure to respirable coalmine dust, with differential estimated effects of these characteristics according to attained category of CWSP at the start of the five-year periods, and the carbon content of the coal being mined. Attained category of CWSP, which is itself a consequence of exposure to dust, was the major predictor of final radiological category. These
estimates of five-year transition probabilities were then applied eight successive times to the changing experience of a hypothetical mining population, entering the industry at age 18 years without prior dust exposure and so with Category 0 CWSP, and exposed throughout the 40 year period to concentrations of 1 mg/m³, 1.5 mg/m³ etc. respirable dust from coals of various percent carbon. The probabilities (%) of showing CWSP Categories 0, 1, 2 and 3, or PMF, at the end of each of the eight successive five year periods (i.e. at ages 23, 28, 33 etc., up to age 58) following exposure to various dust concentrations and for coals of three different % carbon, were estimated in this way, results being presented in a series of tables. We reproduce the results corresponding to an average dust concentration of 2 mg/m³ which were considered the most relevant to the West Burton implementation of the present project (Table 6.3). They estimate, for example, that in coalmines with coal of 86.2% carbon content, 1.19% of miners exposed for 40 years from age 18 years to 2 mg/m³ respirable coalmine dust will have developed PMF by the end of that 40 year working life. Note that the estimates make no allowance for progression of CWSP, or development of PMF, after cessation of dust exposure i.e. after age 58 years, though this can occur (e.g. Maclaren and Soutar, 1985).

<table>
<thead>
<tr>
<th>Coal rank (% carbon)</th>
<th>Radiological Category</th>
<th>Age</th>
<th>Age</th>
<th>Age</th>
<th>Age</th>
<th>Age</th>
<th>Age</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>23</td>
<td>28</td>
<td>33</td>
<td>38</td>
<td>43</td>
<td>48</td>
</tr>
<tr>
<td>83.0</td>
<td>CWSP 0</td>
<td>99.96</td>
<td>99.71</td>
<td>99.15</td>
<td>98.26</td>
<td>97.02</td>
<td>95.41</td>
</tr>
<tr>
<td></td>
<td>CWSP 1</td>
<td>0.04</td>
<td>0.27</td>
<td>0.75</td>
<td>1.47</td>
<td>2.42</td>
<td>3.58</td>
</tr>
<tr>
<td></td>
<td>CWSP 2</td>
<td>0.00</td>
<td>0.02</td>
<td>0.09</td>
<td>0.23</td>
<td>0.46</td>
<td>0.78</td>
</tr>
<tr>
<td></td>
<td>CWSP 3</td>
<td>0.00</td>
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<td>0.00</td>
<td>0.01</td>
<td>0.02</td>
<td>0.05</td>
</tr>
<tr>
<td></td>
<td>PMF</td>
<td>0.00</td>
<td>0.00</td>
<td>0.01</td>
<td>0.03</td>
<td>0.08</td>
<td>0.19</td>
</tr>
<tr>
<td>86.2</td>
<td>CWSP 0</td>
<td>99.96</td>
<td>99.64</td>
<td>98.95</td>
<td>97.83</td>
<td>96.26</td>
<td>94.20</td>
</tr>
<tr>
<td></td>
<td>CWSP 1</td>
<td>0.04</td>
<td>0.32</td>
<td>0.91</td>
<td>1.78</td>
<td>2.92</td>
<td>4.31</td>
</tr>
<tr>
<td></td>
<td>CWSP 2</td>
<td>0.00</td>
<td>0.03</td>
<td>0.12</td>
<td>0.33</td>
<td>0.65</td>
<td>1.09</td>
</tr>
<tr>
<td></td>
<td>CWSP 3</td>
<td>0.00</td>
<td>0.00</td>
<td>0.01</td>
<td>0.02</td>
<td>0.04</td>
<td>0.08</td>
</tr>
<tr>
<td></td>
<td>PMF</td>
<td>0.00</td>
<td>0.00</td>
<td>0.01</td>
<td>0.05</td>
<td>0.14</td>
<td>0.31</td>
</tr>
<tr>
<td>89.0</td>
<td>CWSP 0</td>
<td>99.96</td>
<td>99.58</td>
<td>98.74</td>
<td>97.38</td>
<td>95.45</td>
<td>92.93</td>
</tr>
<tr>
<td></td>
<td>CWSP 1</td>
<td>0.04</td>
<td>0.38</td>
<td>1.06</td>
<td>2.08</td>
<td>3.40</td>
<td>5.01</td>
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<tr>
<td></td>
<td>CWSP 2</td>
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<td>0.04</td>
<td>0.17</td>
<td>0.44</td>
<td>0.87</td>
<td>1.45</td>
</tr>
<tr>
<td></td>
<td>CWSP 3</td>
<td>0.00</td>
<td>0.00</td>
<td>0.01</td>
<td>0.03</td>
<td>0.06</td>
<td>0.12</td>
</tr>
<tr>
<td></td>
<td>PMF</td>
<td>0.00</td>
<td>0.00</td>
<td>0.02</td>
<td>0.08</td>
<td>0.22</td>
<td>0.49</td>
</tr>
</tbody>
</table>

Most of the individual coefficients as estimated in the logistic regression models of Hurley and Maclaren (1987) are highly significant statistically. Most notably the relative risk of some radiological progression (to CWSP 1 or 2 or 3 or PMF) from Category 0, estimated at 1.45 for a doubling of dust exposure, has an associated t-value of 30. Hurley and Maclaren do not however
give standard errors for the estimated transition probabilities, nor for the final estimated prevalences; so it is not possible to give conventional lower and upper values, as is done elsewhere in this project. There are other factors known or believed to influence occurrence of CWSP or PMF, and not taken into account in the relationships derived by Hurley and Maclaren (1987). Most notable of these is physique (Maclaren et al, 1989), important in assessing individual risks, but ignorable in the context of population estimates. In addition, there is evidence that the development of CWSP may be related also to residence time of dust in the lung; but these variations can be ignored in the context of the present project, cumulative exposure being the dominant factor. Similarly, the quartz content of respirable coalmine dust is related both to rapid progression of simple pneumoconiosis and to occurrence of some types of PMF. Both rapid progression of simple pneumoconiosis, and quartz-related increase in types of PMF, are however rare, and so can for present purposes be contained within the general relationships linking cumulative exposure to mixed coalmine dust with CWSP and PMF given below. It would be dangerous however to extrapolate to coalmine dusts with higher than, say 10% quartz on average in the respirable airborne fraction. Smoking habit is unrelated to presence of the small rounded opacities typical of CWSP (Jacobsen et al, 1977); and to development of PMF (Maclaren et al, 1989).

Respiratory symptoms: exposure-response relationships

Analyses of respiratory symptoms in PFR data have focused primarily on chronic bronchitis, understood as both cough and production of phlegm ‘on most days for as much as three months in the year’ (Rae et al, 1971; Marine et al, 1988; Collins et al, 1988). However, Collins et al (1988) distinguish between miners who report breathlessness, i.e. who say that they are breathless when they walk at their own pace on level ground, and those who report symptoms of chronic bronchitis, but without breathlessness. Their results are based on a stratified sample of 895 men from 3,600 miners who had attended four, five yearly surveys at 10 British coalmines, the stratification being to ensure an approximately uniform distribution of age (mean 49 years) and cumulative dust exposure (mean 200 gh/m$^3$) over ranges of interest. Logistic regression methods were used to examine prevalence of respiratory symptoms at the final series of surveys (1968-73) in relation to age, smoking habit, cumulative dust exposure, and presence of small rounded or small irregular opacities on the chest radiographs.

For breathlessness defined as a positive answer to the question ‘Do you have to walk slower than other people on level ground because of your chest?’, the (adjusted) effect of 100 gh/m$^3$ cumulative exposure to dust on the log odds of a positive response was estimated as 0.418 (SE 0.094), giving lower and upper estimates of 0.324 and 0.512 respectively. Differentiating the log odds, we linearise the function as:

$$dp = 0.418 p (1-p) d (100 \text{ gh/m}^3)$$

Furthermore, coefficients in Table 11 of Collins et al (1988) imply that the baseline odds of reporting breathlessness in a population of miners aged 58 years, without CWSP and with zero dust exposure, of whom 80% are smokers and 20% non-smokers, are 0.200. This corresponds to a baseline probability of 16.67%. Substituting these baseline probabilities, we get the estimates of incremental increase in the probability of breathlessness at age 58 years, shown below:
Collins et al (1988) reported a % change in probability of breathlessness at age 58 years of low 4.5, mid 5.8, and high 7.1 per 100 gh/m$^3$ of dust exposure. For symptoms of chronic cough and phlegm, both for more than three months of the year, but without breathlessness, the adjusted logistic regression coefficient per 100 gh/m$^3$ cumulative dust exposure was 0.257 (SE 0.098), giving lower and upper estimates of 0.159 and 0.355 respectively. Again, we linearise by differentiating the log odds, to give the central estimate of incremental change in probability as

$$dp = 0.257 \ p \ (1-p) \ d \ (100 \ \text{gh/m}^3)$$

The baseline odds of a positive response, in a non-dust-exposed population aged 58 years without small opacities and with 80% smokers, 20% non-smokers, are estimated as 0.243, corresponding to a baseline probability of 19.6%. Substituting, these baseline probabilities we obtain an estimate of incremental increase in probability of chronic cough with phlegm (and without breathlessness) at age 58 years shown below:

| Collins et al (1988) | % change in probability of chronic cough with phlegm (and without breathlessness) at age 58 years | Low 2.5 | Mid 4.0 * | High 5.6 per 100 gh/m$^3$ dust exposure |

Note that by definition, these are different men from those who report breathlessness, i.e. the outcomes are additive. These results are broadly consistent with those from the larger study of Marine et al (1988), who considered 3,380 face workers from 20 British coalmines who attended medical surveys 1963-68 when aged 25-64 years. Marine et al give a comprehensive analysis of chronic bronchitis (irrespective of breathlessness), but do not consider breathlessness. Hence, following Jacobsen (1990), we use the Collins et al (1988) estimate.

### Other morbidity of coalminers

There is evidence that coalminers may suffer from occupational skin disorders, from noise-induced hearing loss, from work-related back pain, and possibly for other musculoskeletal conditions. It is difficult however to obtain reliable quantitative data, suitable for the purpose of the present project, on the extent to which these conditions are work-related; and so no attempt is made to provide quantitative estimates for the present project. It is possible however that back pain and other musculoskeletal conditions, rather than the additional dust related lung diseases, are now the major source of occupational morbidity in coalminers.

#### 6.2.4 Implementation of occupational disease rates for the Coal Fuel Cycle

For the UK reference environment, the relevant productivity of the British coal industry in 1990/91 was 1070 te/man-year. All occupational health effects can thus be derived from normalising power station demand against this average production rate; for example, the West Burton B power station would use 4.36 Mte of coal per year, thus the effort required to produce
one year’s supply of coal is 4075 man-years (rounded to 4,000 for implementation). This is likely to be an over-estimate as coal industry productivity will continue to be improved over the prospective lifetime of the power station.

Further to this, it estimated that about 25% of these miners would be employed as face workers; about 50% employed elsewhere underground; and the rest employed on the surface. Miners will of course transfer between these locations in the course of a working lifetime.

The dust standards at British coalmines require that concentrations of respirable coalmine dust be no more than 7 mg/m$^3$ at the designated 70m sampling point in the return roadway. The considerations described by Chamberlain et al. (1971) suggest that this was in practice equivalent to an upper limit of just under 4 mg/m$^3$ for coalminers working along the coalface, a relationship which may have altered because of changes in working practice subsequently. On average, dust concentrations for those working elsewhere underground or on the surface are likely to be much lower. For purposes of the UK implementation, it was assumed that face workers, workers elsewhere underground, and surface workers are exposed respectively on average to 4.5, 1.5 and 0.5 mg/m$^3$ respirable coalmine dust. These figures are likely to be on the higher rather than on the lower side of what is really the case. They imply that the mining population of 4,000 men is exposed on average to 2 mg/m$^3$ respirable dust. This seems a reasonable average value to apply over the working lifetime of an individual miner, given the likely range of jobs he will carry out.

The carbon content of the coal being mined is expected to be 84.8% (dry mineral free basis). We assume that the quartz content of the respirable dust will remain similar to that experienced generally in British coalmines in recent years; i.e. usually below 10% and not above 15% for any sustained periods.

We have been unable to obtain specific measurements of radon daughter levels from the reference coal mining region. Consequently, we use general data from O’Riordan (1991) who gives a summary statistic of 0.1 WLM on average for coal mining in Britain through the 1980s. This value is based on data from what O’Riordan describes as a comprehensive exercise involving over 200 coalmines, both deep and shallow, and so it seems a reasonable basis for estimation. Again at the risk of over-estimation, we applied these exposures to all miners in the reference population, including surface workers.

Another relevant issue is the duration of exposure of individual miners, or equivalently, the turnover of workers within the industry. For computational ease, we base calculations on the unrealistic assumption that the mining population consists of men who start employment at age 18 or 20 years, without significant previous exposures in dusty industries, and who continue working as coalminers over a full 40-year working life. In practice, of course, the mining population at any time is much more diverse in terms of age and coal mining experience, and it is most highly unlikely that many of those employed at any time will remain a full 40 years in the industry. The assumption that 40 man years will be provided by one man working for a full 40 years rather than (say) two miners for 20 years each will tend to over-estimate rather than to under-estimate the overall health impacts.

**Coalworkers’ mortality from lung cancer, and exposure to radon**
The BEIR IV estimates were used for men exposed to 0.1 WLM/yr, whose exposure began at age 20 years, and who remained exposed over a 40-year working life, i.e. until age 60 years. The central estimate of years of life lost to lung cancer is 0.03 years per individual, equivalent to 120 years lost in total in a mining population of 4,000 men, attributable to their working life exposures to radon in coalmines. This includes years lost after retirement from coal mining.

The BEIR IV results also suggest a lifetime risk of lung cancer of 0.069 following occupational exposure to 0.1 WLM/yr radon from age 20 to age 60 years compared with a background risk of 0.067 in the population (US 1980-84) from which the risk estimates were derived. This implies an incremental risk of 0.002, for example corresponding to 8 additional lung cancer deaths in a mining population of 4,000 men.

Together, these results suggest a central estimate of 8 additional lung cancer deaths, with an average of 15 years of life lost for each of these 8 individuals over the 40 years working life of the power plant. This is equivalent to 0.2 deaths, or 3 years of life lost, annually over the period. These deaths are valued using the Value of Statistical Life (2.6 MECU). The BEIR IV model does not give conventional lower and upper bounds on these risk estimates.

**Coalworkers’ simple pneumoconiosis (CWSP), progressive massive fibrosis (PMF), and exposure to respirable coalmine dust**

We assume a coal mining population which enter the industry at age 18 years, and work for 40 years exposed throughout to a dust concentration of 2 mg/m$^3$ respirable coalmine dust, and not consistently exposed to unusually high quartz concentrations. As central estimates of the numbers of men showing PMF or CWSP sufficiently severe to obtain compensation in Britain, i.e. Category 2 or 3 CWSP, we use the results of Hurley and Maclaren (1987) that refer to coal of 86.2% carbon; i.e. slightly higher than the 84.8% carbon content of the reference environment coal, and so leading to slightly higher estimates of risks. For completeness we also give the expected impacts based on coal of 83% and of 89% carbon, respectively. For the mining population of 4,000, the estimated numbers of miners showing PMF or CWSP 2 or 3 after 40 years are shown in Table 6.4.

Assuming a diverse mining population, so that the occurrences are spread evenly throughout the 40 year period of the power plant, the annual figures per year and per TWh can be calculated.

Note that while in principle further occurrences of advanced CWSP or of PMF might be expected in this population after retirement, these estimates as they stand are already unrealistically high when compared with the prevalence data published in recent British Coal Medical Service annual reports. The comparisons are not strictly appropriate, of course: the present estimates refer to prevalences at the end of a 40-year working life, and men who meet this criterion are at best a very small proportion of current miners.
The data corresponding to a carbon content of 86.2% are the central estimates for the purposes of the present evaluation. As noted earlier, Hurley and Maclaren’s (1987) report does not give data from which lower and upper bounds may be estimated. It may be tenable to use the values corresponding to 83% and to 89% carbon content as surrogate ‘lower’ and ‘upper’ values respectively. Our subjective impressions are that they may be of the right orders of magnitude; but it must be understood that logically, they have been derived on a fundamentally different basis from the lower and upper bounds estimated elsewhere in this project.

In order to derive preliminary damages for PMF, we have estimated the proportion of miners for which pneumoconiosis is the primary cause of death. No attempt has been made to quantify health endpoints other than death and the damage costs produced are therefore likely to be an underestimate of actual impacts.

Published data (Miller et al, 1981) for miners surveyed in the 1950s, with a mortality follow-up over the subsequent 22 years, revealed 21% of the men studied had pneumoconiosis registered as the cause of death. Therefore, these values are applied to the central estimates and the Statistical Value of Life (2.6 Million ECU) is used to quantify such a death from chronic disease.

**Respiratory symptoms of chronic bronchitis, and long-term exposure to respirable dust**

For the mining population, exposed, on average, to 2 mg/m$^3$ respirable dust, for 40 years from age 18 to age 58 years and furthermore, as in Hurley and Maclaren (1987), assuming that a year’s coal mining work is equivalent to 1,631 hours exposed occupationally. Then 40 years’ work is associated with a cumulative exposure of 130 gh/m$^3$ respirable dust exposure.

We do not have recent data on what proportion of coalminers are lifelong non-smokers at age 58 years, nor do we have a basis for estimating how this proportion may change over the coming 40 years. We assume for purposes of implementation that 20% of those employed are lifelong non-smokers, the remaining 80% being smokers.

Finally, because estimates refer to miners without small opacities, we draw on results from the previous section to estimate that the population at risk at age 58 years is 3,600 men without CWSP.

For breathlessness, as noted earlier the base-line probability of reporting breathlessness, without dust exposure, is estimated as 16.67% (odds 0.20). The central, lower and upper relative risks, corresponding to 130 gh/m$^3$ cumulative exposure, are estimated as 1.72, 1.52 and 1.95.
respectively based on Collins *et al* (1988). Thus the central, lower and upper estimates of prevalence of breathlessness in dust exposed miners without CWSP after 40 years work at 2 mg/m$^3$ respirable dust are 25.60%, 23.31% and 28.06%. Thus the estimates of incremental percentage increases against a baseline prevalence of 16.67% are 8.93%, 6.64% and 11.39% respectively.

Applied to a population of 3,600 men at risk, these figures give central, lower and upper estimates of dust-related increase in breathlessness of 321, 239 and 410 respectively over 40 years. Thus the estimates for annual increments, assuming that these occur smoothly over the 40 year period in a mixed-aged mining population, are 8, 6 and 10.25 men per annum, respectively.

For chronic cough and phlegm, without breathlessness, and using similar population data, the baseline probability of reporting chronic cough and phlegm, without breathlessness, at age 58 years, is 19.6%. The relative risks estimated from Collins *et al*, after exposure of 130 gh/m$^3$, are 1.40, 1.23 and 1.59 as central, lower and upper estimates respectively. These imply prevalences of 25.4%, 23.0% and 27.9% respectively after 40 years exposure at 2 mg/m$^3$ respirable dust, equivalent to increments of 5.8%, 3.4% and 8.3% against the background prevalence of 19.6%.

Applied to a population at risk of 3,600 men, these percentage increments imply respectively 209, 122 and 299 dust related cases at the end of 40 years. Assuming these impacts are spread evenly in time in an age-mixed population at risk, we get central, lower and upper estimates of annual numbers of new cases of chronic cough and phlegm, without breathlessness, due to dust in miners aged 58 years without CWSP, of 5.2, 3.1 and 7.5, respectively.

For both breathlessness and chronic cough and phlegm, no attempt has been made to value the incremental health endpoints. This is highlighted as a priority area warranting further work.

*Implementation for the German reference mine*

For the German implementation, the same functions were applied. A considerable decrease in pneumoconiosis mortality has been observed over the last thirty year in Germany. According to several experts, no new cases of pneumoconiosis will occur in the German mining industry because of higher safety standards (dust reduction) and medical provisions (transfer to a work in reduced dust conditions). Therefore, no additional cases of coalworkers’ pneumoconiosis due to the reference Lauffen power station was assumed as a lower boundary. However, as pneumoconiosis is a progressive disease that might develop even if the initial exposure is stopped, an attempt was made to estimate an upper boundary of possible effects by using the dose-response relationship derived by (Hurley and Maclaren, 1987).

The dust standards at German underground coalmines require that concentration of respirable coal dust is less than 4 mg/m$^3$ (quartz concentration < 4%). In accordance with the approach of the UK analysis, an average exposure of 2 mg/m$^3$ respirable dust was assumed, applied over the working lifetime (40 years) of an individual miner.

The relevant productivity of the German coal mining industry is 501 te/man-year (the average over the period 1986-1990), leading to an effort of 1,656 man-years to produce one year’s coal supply of the Lauffen power station. As the analysis of (Hurley and Maclaren, 1987) is based on
an individual’s working life of 40 years, the effort of 1,656 man-years translates into an effort of 41.4 ‘working lives’. The carbon content of the coal from the Ruhrgebiet is in the range of 82% - 92% (dry and ash-free) with an average of about 88%, so that the results of (Hurley and Maclaren, 1987) (Table 6.4) referring to a carbon content of 89% are used.

Exposure to radon and to radon daughters leads to an increased risk of lung cancer in underground coalminers. The estimation of additional deaths due to mining activities in the German reference environment is also based on the BEIR IV estimates.

Müller et al. (1992) estimate an average effective equivalent dose of 1.2 mSv/year from exposure to radon and its daughters for the Ruhrgebiet coalfields. Although this value was derived during the development of a measuring system and not for the purpose of risk assessment, it is used in the present analysis although it will lead to a considerable overestimation of effects. An ongoing epidemiological study on lung cancer risk among coalminers carried out by the University of Cologne might provide better risk figures in the future.

Using a conversion factor provided by O’Riordan (1991) (10 mSv = 1 Working Level Month) the exposure of 1.2 mSv/year is equivalent to 0.12 WLM/year. To give a range of possible effects, the BEIR IV estimates derived for exposure rates of 0.1 and 0.2 WLM/year are used. Assuming an exposure beginning at the age of 20 and continue until the age of 60, the BEIR IV results suggest a lifetime risk of lung cancer between 0.069 and 0.072. Taking into account the baseline risk of 0.067 (US 1980 - 84), the incremental risk due to mining activities is between 0.002 and 0.005, i.e. 2 to 5 cases of lung cancer have to be expected in a population of 1000 workers.

Besides the lifetime risk, BEIR IV provides an estimation of Years of Life Lost (YOLL) leading to an average of between 0.03 and 0.07 YOLL per individual worker from an exposure of 0.1 and 0.2 WLM/year respectively.

Taking into account the productivity of 501 te/man-year in the German reference environment and assuming a working life of 40 years and a similar exposure to all workers - assumptions that obviously lead to an overestimation of effects - the expected health impacts were calculated.

6.2.5 Occupational disease in other activities of the fuel cycles

The level of detail shown above for the analysis of occupational disease has not been achieved in other industrial sectors. To a large degree, this is due to the presence of more historical data and the well characterised and severe health endpoints associated with coalminers. The picture is less clear for other fuel cycles activities; in cases, some information can be obtained from occupational disease statistics, though these are not likely to provide the same level of accuracy as the above method. Nonetheless, such data does allow preliminary estimates of impacts and can highlight the priority areas which warrant development of a more impact-pathway based approach. In all cases, work is need both in the primary research into the incidence of disease and for methods to place economic values on any non-fatal health endpoints. Possible areas include:
• Long-term exposure to diesel fumes (offshore exploration, construction activities, etc.);
• Hearing loss through occupational noise;
• Long-term exposure to chemicals, including drilling muds from offshore drilling operations;
• Long-term occupational disease associated with deep sea diving;
• Occupational exposure to particulates (power station workers, construction workers, etc.);
• Musculo-skeletal conditions.

6.3 Occupational Health Effects from Accidents

6.3.1 Introduction

Mortality and morbidity due to accidents are impacts which are best analysed without the use of the damage-function approach. The methodology presented here follows the implementation for the German and UK reference environments for various fuel cycles. However, slight differences in the implementation between the two countries are inevitable, in so far as the statistics available in the UK and Germany are categorised differently.

Data on occupational health are rarely available for specific facilities. Moreover, the accident rate in any individual facility will not be statistically significant unless a large number of years of data is used. Instead, aggregated data representing average accident frequencies in the relevant sectors are used throughout the fuel cycle analysis. Such estimates have a higher level of confidence attached to them, as they more accurately reflect average accident rates.

In producing risk estimates based on statistics, care must be taken in deciding how far back data should be considered. If data are used which are ten or more years old, they are unlikely to be representative of modern conditions with improved safety standards. However, if only the latest data are used, for example for less than the last five years, the short time span will mean results are susceptible to short term fluctuations, for example created by major incidents. Within the ExternE Project, five years is regarded to be an appropriate time span in which figures still remain representative of modern conditions. As far as possible, average values taken from accident statistics over the five year time span 1986 - 1990 have been used in assessments to date, though the German data has recently been updated to cover the period 1988-1992. For future studies, with updated reference years, further updates will be required.

6.3.2 Categories of health risk estimates in the UK and Germany

The categories of accidental injury in the UK conform to the UK Reporting of Injuries Diseases and Dangerous Occurrences Regulations 1985 (RIDDOR), under which occupational accidents have been reported since April 1986. Three categories are distinguished as follows (BCC, 1989):
• Fatalities;
• Major injuries, defined to include major fractures, amputation, serious eye injuries, some causes of loss of consciousness and any injury requiring hospital treatment for more than 24 hours;
- Minor injuries, defined to include other accidents responsible for the loss of more than three working days.

For transport accidents, the categories are similar. Major injuries are generally those requiring a person to be detained in hospital, whereas minor injuries are those requiring other medical attention. The reporting is done by the police on the basis of observation at the scene of the accident, rather than by medical examination (UKDTP, 1991). For the purposes of this project, these categories of major and minor were assumed to be the same as for RIDDOR.

The assessment of occupational health impacts from the German fuel cycles are mainly based on statistical data from the German employees’ insurance system (with the exception of coalminers described above), reported by the ‘Hauptverband der gewerblichen Berufsgenossenschaften’ (Employees’ Industrial Compensation Society). Data are categorized according to the German ‘Reichs Versicherungs Ordnung’ that defines occupational accidents and diseases as follows:

- **Occupational accidents** lead to the inability to work for more than three days. All such accidents have to be reported (reported accident) though the term ‘occupational accident’ is not specifically defined. According to current jurisdiction, an accident is a sudden, body damaging external impact. Its sudden appearance is the main difference from occupational diseases;

- **Occupational diseases** also lead to the inability to work for more than three days and must also be reported (reported disease);

- Permanent disabilities are classified as a certain percentage of **reduction in earning capacity**;

- Occupational accidents and diseases that have to be compensated by a pension, compensation or death benefit are classed as **compensated accidents / diseases**. Accidents and diseases have to be compensated if they cause a **reduction in earning capacity of at least 20%**;

- **On-road accidents** (journeys to and from the working place) are regarded as **occupational accidents**.

- An **occupational disease** is regarded as **cause of death** if the disease is recognized as the main cause of death or if the person had been compensated previously because of a reduction in earning capacity of more than 50%.

Using this classification, occupational health statistics for the German reference environment are classified into:

- Number of deaths caused by occupational accidents;

- Number of deaths caused by occupational diseases;

- Years of Life Lost (YOLL) caused by fatal accidents and diseases;

- Major non-fatal occupational accidents;

- Major non-fatal occupational diseases;

- Minor non-fatal occupational accidents;

- Minor non-fatal occupational diseases.

Accidents and diseases leading to a Reduction in Earning Capacity of at least 20% (compensated accidents and diseases) are considered as major accidents and diseases, whereas not-compensated accidents and diseases are regarded as minor. Due to country specific differences between the relevant statistics, the definition of major and minor impacts in Germany is not
equivalent to the UK definition, but it has been assumed that the severity of the impacts are comparable for subsequent valuation.

6.3.3 Assessment of occupational health effects

From the fuel cycles selected, the following phases have been identified for which occupational accidents or disease may arise:

Extraction:
- Coal mining;
- Lignite mining;
- Offshore activity, including exploration, development and extraction of oil and gas;
- Limestone extraction.

Transportation:
- Coal transportation;
- Limestone transportation;
- Lignite transportation
- Gas transportation (pipeline);
- Oil transportation (pipeline or tanker);
- Transportation of personnel (including power station related road traffic);
- Waste material transportation.

Processing and generation:
- Oil refining;
- Gas treatment;
- Power station operation.

Other:
- All plant construction, including power plants;
- All plant and structure decommissioning;
- Transmission.

We have not discussed individual activities associated with renewable technologies here - the details are given in the individual fuel cycles concerned. For most renewable technologies, the greatest occupational risks are from the construction phases. Because of this, some renewable assessments include more detail on construction activities, for example, for the wind fuel cycle, data has been analysed on occupational accidents from manufacture of materials.

The potential radiological impacts to occupational workers was considered in the previous Chapter and is not repeated here. All other occupational accidents from the nuclear fuel cycle were presented in the nuclear fuel cycle (European Commission, 1995c).

The stages listed above are considered individually below. Each summarises the approach used and the source and quality of the statistical information available in the analysis for the countries in which implementation has been performed. For a full discussion of each activity, the reader should refer to the original fuel cycles.
6.4 Occupational Accidents from Extraction

6.4.1 Coal mining

Until very recently, UK coal mining was dominated by a single company (British Coal Corporation) and therefore mines work to common safety procedures. It is therefore believed to be justified to use data from the whole of the UK industry (BCC, 1989). Data on fatalities is the annual average from the four year period 1985-89, whilst the more numerous non-fatal accidents are those from the last year of the period alone. Using UK Government data on coalmine output (UKDUKES, 1991), the specific accident rates are shown in Table 6.5.

For Germany, more than 90% of hard coal is produced in the regions of the Saarland and the Ruhrgebiet coalfields; the German implementation thus used occupational health statistics for the German hard coal industry as a whole. Direct health impacts from fatal and non-fatal accidents and diseases are estimated from the statistical data of the Bergbau Berufsgenossenschaft (mining employees’ compensation society) (BG Bergbau, 1991), shown in the Table below:

Table 6.5 Occupational accidents from coal mining.

<table>
<thead>
<tr>
<th>Accident category</th>
<th>UK accident rate/Mill. tonnes</th>
<th>Accident category</th>
<th>German accident rate/Mill. tonnes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Killed</td>
<td>0.20</td>
<td>Killed</td>
<td>0.55</td>
</tr>
<tr>
<td></td>
<td></td>
<td>YOLL</td>
<td>20.6</td>
</tr>
<tr>
<td>Major injury</td>
<td>8.14</td>
<td>Major accident/disease</td>
<td>16.3</td>
</tr>
<tr>
<td>Minor injury</td>
<td>69.96</td>
<td>Minor accident/disease</td>
<td>164</td>
</tr>
</tbody>
</table>

6.4.2 Limestone extraction

Accident rates occurring at UK quarries are collated by the Health and Safety Executive. Annual reported accidents are recorded for the sector, though the rates are from all quarrying activity and are not specific to limestone extraction. A number of occupational diseases do occur in the operational quarry workforce. The two major reported illnesses are Pneumoconiosis and Vibration White Finger. However, incidence is extremely low; for the 76,000 employees in the UK quarry industry, only 2 cases of Pneumoconiosis and 16 cases of Vibration White Finger were reported over the years 1989 - 1992.

Occupational health risks from German limestone extraction are estimated on the basis of occupational health statistics of the quarry workers’ compensation society. As there are no data available on the limestone productivity, productivity figures expressed as man-years/tonne are taken from the hard coal industry, taking into account that this assumption is likely to lead to an overestimation of impacts.
Table 6.6 Occupational health impacts from limestone extraction (Germany).

<table>
<thead>
<tr>
<th>Accident category</th>
<th>Accident rate /million tonnes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Killed</td>
<td>0.37</td>
</tr>
<tr>
<td>YOLL</td>
<td>10.5</td>
</tr>
<tr>
<td>Major accident/disease</td>
<td>7.7</td>
</tr>
<tr>
<td>Minor accident/disease</td>
<td>201.8</td>
</tr>
</tbody>
</table>

6.4.3 Lignite mining

The German evaluation of lignite mining from open-cast mining fields are estimated from the statistical data of the Bergbau Berufsgenossenschaft (mining employees’ compensation society). As the accident statistics only provide aggregated risk figures for the whole German coal industry, national average values are used for risk assessment. Although the basic statistics do not differentiate between underground hard coal mining and open cast lignite mining, the total number of compensated cases is reported for each sector. Assuming that the same ratio is a reasonable indicator for the difference of fatalities, this ratio is used to estimate the proportion allocated to open cast mining activities from the number of total fatalities in the German coal industry. In contrast to underground hard coal mining, there are no cases of silicosis observed in Germany resulting from open cast lignite mining activities.

Table 6.7 Occupational health impacts from lignite extraction (Germany)

<table>
<thead>
<tr>
<th>Accident category</th>
<th>Accident rate /million tonnes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Killed</td>
<td>0.0065</td>
</tr>
<tr>
<td>YOLL</td>
<td>0.24</td>
</tr>
<tr>
<td>Major accident/disease</td>
<td>0.20</td>
</tr>
<tr>
<td>Minor accident/disease</td>
<td>3.4</td>
</tr>
</tbody>
</table>

6.4.4 Offshore oil and gas activity

The extraction of oil and gas involves a number of separate phases, which can be broadly categorised into oil and gas exploration and appraisal; oil and gas development; and oil and gas extraction. For the UK and German implementations, all oil and gas extraction activities have centered on North Sea operations in the UK and Norwegian sectors. The North Sea is a unique working environment; the main factors that contribute to health and safety risks offshore are:

- The offshore environment is hostile and dangerous;
- Workers are exposed to constant risks (e.g. explosive nature of gas, helicopter travel, structural failure of platforms, etc.);
- Some jobs are associated with high risks (e.g. drilling and diving);
• Workers are housed on the work site;
• There may be psychological stress from constant company, long hours and isolation (the living accommodation is sometimes overcrowded and there are limited leisure facilities).

In all cases, accidents may be compounded by the time and distance separating casualties from professional medical help. Of all the phases of offshore work, drilling and diving operations are undoubtedly the most dangerous activities. These are primarily centred around the exploration, appraisal and development stages; in spite of this, medical facilities on drilling rigs are usually not very sophisticated.

**Offshore accidents**

The figures for the UK are taken from Government statistics for offshore occupational accidents within the UK Continental Shelf (UKCS) (DEn, 1991), with information on minor incidents taken from reported injuries at work for oil and gas extraction (CSO, 1993). These figures include: ‘all accidents and dangerous occurrences on or near installations and pipeline works or on attendant vessels in the course of any operation undertaken in connection with an installation. They also include accidents and dangerous occurrences in respect of pipelines or in the course of pipeline works. Accidents involving helicopters flying to or from installations are only included if they occur in the vicinity of the installation.’

Separate accident rates were detailed for drilling activity in the UK sector in the natural gas fuel cycle (European Commission, 1995d), as an indication of the accident rates during the more hazardous exploration, appraisal and development activities.

During the offshore extraction phase, accident rates are considerably lower. The UK accident rates per unit of output are estimated from the sector production and employment figures. As statistics are given with no differentiation between oil and gas activities, accident rates are split according to energy production, based on conversion factors quoted for UK production. Total offshore production (excluding drilling accidents) are normalised against average UK production to produce the values in Table 6.8.

Accident rates in the Norwegian sector are taken from the German oil fuel cycle (European Commission, 1995d), based on data from the Norwegian Petroleum Directorate (1992). The values assume that 5% of accidents (excluding fatalities) are serious and 95% are minor.

**Table 6.8** Occupational accident rates from offshore production.

<table>
<thead>
<tr>
<th>Accident category</th>
<th>UK accident rate/Mill. tonnes</th>
<th>Accident category</th>
<th>Norwegian accident rate/Mill. tonnes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fatal accidents</td>
<td>0.027</td>
<td>Fatal accidents</td>
<td>0.0036</td>
</tr>
<tr>
<td>Serious accidents</td>
<td>0.388</td>
<td>Serious accidents</td>
<td>0.34</td>
</tr>
<tr>
<td>Minor accidents</td>
<td>2.713</td>
<td>Minor accidents</td>
<td>6.55</td>
</tr>
</tbody>
</table>
It should be noted that the UK and Norwegian reporting systems are slightly different; in addition, the Norwegian results above include all offshore activities (including drilling activity). The results do not include deaths and injuries from the Piper Alpha explosion, as these should not be considered in a ‘normal operational risk’ assessment. Furthermore, they exclude a number of other serious accidents, such as helicopter crashes, fires and structural failures. These are considered separately in Chapter 15 in major accident assessment. In all offshore analysis, some care must be taken in the analysis of recent statistics because of the recommendations of the Cullen Report into the Piper Alpha disaster (DEn, 1990). These recommendations have lead to significant changes in the design and management of new and existing facilities (Taylor, 1991).

**Offshore occupational disease**

The traditional boundaries between occupational health, public health and clinical medicine tend to be somewhat blurred when considering operations in the oil and gas industry. The often isolated and exceptional conditions under which operations are carried out dictate that occupational health must extend beyond the normal scope into the living environment, including psycho-social factors, as well as specific occupational activities such as diving. The analysis of such impacts is difficult as little published information is available. The industry workforce tends to be young, fit and comprise mostly of contractors. Workers thus tend to be itinerant, making data collection and analysis difficult.

Some indication of the level of offshore work related diseases can be taken from Norwegian data (NPD, 1993). These show the additional diseases recorded that may be the result of occupational factors. Cases reported by category were presented in the gas fuel cycle (European Commission, 1995d). Muscular and skeletal conditions are the most common complaint usually resulting from physical strain. Another major group is ear diseases, including noise-induced hearing loss, often caused by chronic exposure in the present or former work. The third major group is skin disease, particularly eczema due to exposure to various chemicals. The group is dominated by employees with hand eczema after contact with drilling muds.

However, no attempt has been made to value these effects; it can be assumed that most are of minor importance as they represent short-term effects. There is little information on the potentially greater effects from long-term diseases related to offshore work. There are some obvious hazards which could lead to health effects in the longer term (e.g. exposure to noise, drilling muds, stress, etc.). The most important of which include occupational exposure to carcinogenic pollutants and long-term effects of diving activity. Further collection of health statistics in this area is important, as it will determine whether a full impact pathway approach is warranted. Such data is therefore seen as a research priority.

### 6.5 Occupational Accidents from Transportation

In contrast to nuclear-energy systems and many of the renewable energies, the volume of fuel to be transported is rather extensive when using fossil energy sources. To avoid repetition, the
following section is divided by transport mode rather than by fuel. There is however an additional complication with transport; the potential impact of transport accidents on the general public. Although strictly speaking this is a public health issue, as a similar methodology is appropriate we have included data here in the discussion of occupational transport accidents.

6.5.1 Rail Transportation

For the UK, rail is the dominant mode of transportation for coal and limestone to power stations. Published statistics for accidents in the UK rail industry (UKDTP, 1991) do not distinguish between passenger and freight related accidents. It is therefore necessary to make some arbitrary assessment to divide accidents between the two:
- It is assumed that all accidents to passengers are related to passenger transport;
- For accidents to railway staff, it is assumed that the accident frequency is proportional to the train km, and therefore that 23% are related to freight transport;
- For accidents to other people (neither passengers nor staff), it is assumed that accidents due to train crashes are proportional to distance travelled, and therefore that 23% are due to freight. For other accidents related to train movements and on railway premises, it is assumed that freight loading, unloading and shunting is the cause, and therefore all accidents are due to freight.

For accidents due to train crashes, the accident probability is calculated per train km from UK statistics. It is assumed that these are representative of coal freight trains. For other accidents on railway premises, the accident rates are calculated per tonne of freight moved. These train crash results can then be converted to fuel cycle impacts based on the annual transport km involved. All the accidents are classified as occupational, whether they relate to railway staff or other, except those to non-railway staff in train crashes.

In Germany, about one third of the coal is moved by rail and the rest by barge. Because railroad transportation statistics for accidental deaths and injuries do not distinguish between the types of freight carried, it is not possible to determine the effects of transporting coal by rail (Stat. Bundesamt, 1991; BV, 1991). The simplest way to estimate this value is to assign a proportionate share of all railroad deaths to coal transport. Regarding the occupational risks this apportionment should be on the basis of the total weight of the material transported, i.e. risks per tonne instead of risks per kilometre travelled, because the majority of accidents are expected to occur during loading, unloading and shunting.

<table>
<thead>
<tr>
<th>Accident category</th>
<th>UK accident rate/Mte freight</th>
<th>Accident category</th>
<th>German accident rate/Mill. tonnes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Killed</td>
<td>0.11</td>
<td>Killed</td>
<td>0.027</td>
</tr>
<tr>
<td></td>
<td></td>
<td>YOLL</td>
<td>0.88</td>
</tr>
<tr>
<td>Major injury</td>
<td>0.43</td>
<td>Major accident/disease</td>
<td>0.31</td>
</tr>
<tr>
<td>Minor injury</td>
<td>5.83</td>
<td>Minor accident/disease</td>
<td>16.9</td>
</tr>
</tbody>
</table>
The calculation of public accidents from coal transportation (UK only) is based on the data described above. The results are shown below.

**Table 6.10** Public accidents from UK freight transportation.

<table>
<thead>
<tr>
<th>Accident category</th>
<th>Accident rate /Mill. train km</th>
</tr>
</thead>
<tbody>
<tr>
<td>Killed</td>
<td>0.015</td>
</tr>
<tr>
<td>Major injury</td>
<td>0.011</td>
</tr>
<tr>
<td>Minor injury</td>
<td>0.038</td>
</tr>
</tbody>
</table>

**6.5.2 Barge transport**

For the German implementation, the same method as coal transportation was applied with the share of the total risks caused by coal transport calculated on the basis of the total weight of the material transported.

**Table 6.11** Occupational health impacts from barge freight transportation (Germany).

<table>
<thead>
<tr>
<th>Accident category</th>
<th>Accident rate/Mill. tonnes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Killed</td>
<td>0.108</td>
</tr>
<tr>
<td>YOLL</td>
<td>3.6</td>
</tr>
<tr>
<td>Major accident/disease</td>
<td>0.61</td>
</tr>
<tr>
<td>Minor accident/disease</td>
<td>15.9</td>
</tr>
</tbody>
</table>

**6.5.3 Road transport**

Increased economic activity at all major stages of a fuel cycle will have a tendency to generate additional road traffic. In principle, it should be possible to calculate the additional journeys generated by each fuel cycle stage and to apply traffic accident statistics to these to calculate the incremental accidents. In practice, it is rather difficult to calculate the additional journeys in many cases. The analysis to date in the UK implementation has concentrated on power stations, which are discreet facilities for which reliable calculations are generally provided.

Additional road journeys include both movements of material by commercial heavy goods vehicles and the daily commuting of workers. For worker transport, determining the average journey length is problematic, but some idea of the level of transport can be drawn from analysis of the local labour market. For example, with the UK coal fuel cycle, it was concluded that most of the workforce would come from within a designated commuting zone, it was therefore possible to derive annual incremental traffic and thus accident rates per year. Estimates of the injury rates due to road travel are taken from Department of Transport statistics (UKDTP, 1991). These accidents will also include members of the public and thus, they are not confined to the occupational workforce alone.
Table 6.12 Road accident injury rates (UK).

<table>
<thead>
<tr>
<th>Accident category</th>
<th>Accidents /Mill. vehicle km</th>
</tr>
</thead>
<tbody>
<tr>
<td>Killed</td>
<td>0.013</td>
</tr>
<tr>
<td>Major injury</td>
<td>0.147</td>
</tr>
<tr>
<td>Minor injury</td>
<td>0.669</td>
</tr>
</tbody>
</table>

The equivalent values used in the German implementation are shown below, calculated on the basis of the transport capacity expressed as ton-kilometers.

Table 6.13 Accident injury rates for freight road transport (Germany)

<table>
<thead>
<tr>
<th>Accident category</th>
<th>Accident rate/10⁹ tonne km</th>
</tr>
</thead>
<tbody>
<tr>
<td>Killed</td>
<td>0.85</td>
</tr>
<tr>
<td>YOLL</td>
<td>29.0</td>
</tr>
<tr>
<td>Major accident/disease</td>
<td>10.9</td>
</tr>
<tr>
<td>Minor accident/disease</td>
<td>343.7</td>
</tr>
</tbody>
</table>

A significant short term impact will arise in power station construction stages, when the number of workers on site is much greater and there are increased materials and goods deliveries. Transport requirements during construction arise from:

- Delivery of bulk building and civil engineering material;
- General construction supplies, services and personnel;
- Delivery of heavy and abnormal loads.

The first of these categories would create the greatest flow of heavy goods vehicles, while the second will be the major influence on overall construction traffic flows. The analysis of heavy goods vehicles is more complicated as it is often difficult to predict average haulage distances, apart from when clear distances can be determined, for instance the distances between power stations and waste disposal sites. It should be stressed that no differential has been applied between heavy goods traffic and worker transportation to date. This is likely to introduce some error into the analysis.

6.5.4 Marine transport (crude oil transport by tanker)

Data on occupational accidents for German marine transport were derived from average figures for deep-sea transportation from data provided by the German employees’ compensation society.

Major accidents in the oil tanker industry are common arising from collisions or from fires; these severe incidents are discussed in Chapter 15.
Table 6.14 Occupational health impacts from crude oil transportation by tanker (Germany).

<table>
<thead>
<tr>
<th>Accident category</th>
<th>Accident rate/Mill. tonnes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Killed</td>
<td>0.16</td>
</tr>
<tr>
<td>YOLL</td>
<td>5.4</td>
</tr>
<tr>
<td>Major accident/disease</td>
<td>0.94</td>
</tr>
<tr>
<td>Minor accident/disease</td>
<td>12.4</td>
</tr>
</tbody>
</table>

6.5.5 Pipeline transportation

Once operational, offshore pipelines only require occasional cleaning (pigging) operations and some routine maintenance. The majority of occupational accidents are considered to be included in the estimates of offshore operation.

The operation of on-shore pipelines requires a small workforce to perform infrequent maintenance and repair operations. As an example, the pipeline studied in the UK natural gas fuel cycle is inspected fortnightly by helicopter, and annually by foot, to identify possible damage. The man-hours generated by these actions, and hence associated accident rates, are extremely low.

6.6 Occupational Health Effects from Process Plant and Power Plant Operation

Occupational accidents in the UK arising from oil refining are taken from accident statistics for the chemical industry (CSO, 1993). These data are also likely to include employees not directly involved in process operations. Occupational accidents arising from the operation of the gas treatment terminal and all fossil fuel power plants in the UK are taken from data presented in CSO (1993) for the energy and water supplies industry. In both cases, the data is not specific to the exact processes under analysis.

Table 6.15 Occupational accidents in the chemical industry and the energy and water supplies industry (UK).

<table>
<thead>
<tr>
<th>Accident category</th>
<th>Accident rate per thousand employees</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Chemical industry</td>
</tr>
<tr>
<td>Killed</td>
<td>0.021</td>
</tr>
<tr>
<td>Serious injury</td>
<td>1.6</td>
</tr>
<tr>
<td>Minor injury</td>
<td>11</td>
</tr>
</tbody>
</table>

For the German implementation, occupational health risks related to crude oil refining are estimated based on the work effort per million tonnes refinery throughput. Data on occupational health provided by the employees’ compensation society are average figures from the German chemical industry.
Table 6.16 Occupational health impacts from crude oil refining (Germany).

<table>
<thead>
<tr>
<th>Accident category</th>
<th>Accident rate/Mill. tonnes refinery throughput</th>
</tr>
</thead>
<tbody>
<tr>
<td>Killed</td>
<td>0.010</td>
</tr>
<tr>
<td>YOLL</td>
<td>0.34</td>
</tr>
<tr>
<td>Major accident/disease</td>
<td>0.28</td>
</tr>
<tr>
<td>Minor accident/disease</td>
<td>10.1</td>
</tr>
</tbody>
</table>

The estimation of power plant operation was carried out on the basis of the number of employees in the public electricity supply of Baden Württemberg. Assuming that all the employees either belong to the occupational group ‘electrical engineering’ or ‘management/services’, health impacts are estimated using the accident statistics of the respective compensation societies. As not all of the employees directly work in the power plant, the results include risks stemming from all public services of the utilities, e.g. the maintenance of distribution facilities, so they should be regarded as an upper estimate of the power plant operation risks. As occupational statistics do not consider the type of the power plant, the results are average values and not for any individual power station.

Table 6.17 Occupational health impacts from power plant operation (Germany).

<table>
<thead>
<tr>
<th>Accident category</th>
<th>Accident rate per person-year</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Electrical engineering</td>
</tr>
<tr>
<td>Killed</td>
<td>3.5 E-5</td>
</tr>
<tr>
<td>YOLL</td>
<td>0.0011</td>
</tr>
<tr>
<td>Major accident/disease</td>
<td>0.0007</td>
</tr>
<tr>
<td>Minor accident/disease</td>
<td>0.027</td>
</tr>
</tbody>
</table>

6.7 Occupational Health Effects from Construction and Decommissioning

6.7.1 Construction

Construction is an economic activity which has one of the highest rates of occupational accidents, especially with respect to fatalities. For any incremental fuel cycle, the most significant construction activity is the building of the power station itself. However, it has not proved possible to obtain statistically significant data specific to power station construction. Data related to accident rates in general construction activities have therefore been used instead.

UK Government statistics (CSO, 1991) give occupational accident rates in each major sector of economic activity, including for construction as a whole. To identify a specific accident rate it is necessary to choose a metric common to all construction activities. Total economic output of the
sector is chosen. The specific accident rates calculated are given below. These are accident rates for construction only, and do not include accidents in other industrial sectors supplying the construction industry.

To convert these rates to the accident related to power station construction, estimates of the capital cost of the station is required; these were given in each individual fuel cycle report. In addition, the operational lifetime of the plant must be known.

Table 6.18 Occupational accidents from construction (UK).

<table>
<thead>
<tr>
<th>Accident category</th>
<th>Accident rate per £ 10^9</th>
</tr>
</thead>
<tbody>
<tr>
<td>Killed</td>
<td>2.32</td>
</tr>
<tr>
<td>Major injury</td>
<td>62</td>
</tr>
<tr>
<td>Minor injury</td>
<td>298</td>
</tr>
</tbody>
</table>

For the German implementation, the construction risks are estimated on the basis of investment figures and of the work effort required, using available accident statistics (Bock-Werthmann, 1986). As there are no specific data on occupational impacts from power plant construction, average health effects in the major industrial activities involved in the power plant construction are estimated. Following this approach, health impacts not only from construction, but also from material supply are included.

The investment costs of the power station are broken down into the main fields of construction, proportional to the direct effects on different industry branches. The manpower related to these investment portions is then calculated using the figure ‘turnover per person’ which is published in the ‘German Statistical Yearbook’ (Stat. Bundesamt, 1991). Accident statistics of the compensation societies (BAS, 1991) are used to evaluate the expected construction risks - fatal and non fatal accidents and diseases - related to the man-power figures. The total risk of the construction phase of the power-plant attributable to material supply and construction activities is related to the total expected output during the lifetime of the power plant.

6.7.2 Decommissioning and dismantling

Because there are no data available on the dismantling of power-plants, decommissioning risks are estimated in the same manner as construction risks were calculated, that is to say on the basis of investment figures and manpower.

Decommissioning information for the German implementation was analysed for the ‘construction’ industrial sector considered. The costs of dismantling are estimated to be 2% of the total investment costs, the other input data are the same as in the case of power-plant construction.
6.7.3 Transmission

Occupational accidents associated with transmission lines will occur both during construction and operation and can be taken from industry statistics. There is however another potential impact; the effects of electromagnetic radiation from transmission lines. This is a controversial area and no definitive answer is possible at the moment. Investigations into the possibility of an occupational hazard of cancer from exposure to extremely low frequency electromagnetic fields have not provided any evidence of a quantitative relationship between risk and level of exposure (NRPB, 1992), though further research is still needed. However, the issue is potentially more important for public health impacts. There is some suggestive evidence of an association between childhood cancer and residential electromagnetic field exposure, though there is currently insufficient evidence to allow firm conclusions to be drawn. The area is currently the subject of much research and warrants close attention in future studies.

6.8 The Concept of Net-Risk

The quantification of occupational health impacts to date within the ExternE Project has been focused on the calculation of gross-impacts, i.e. the total impacts related to any fuel cycle activity is quantified and monetised. Following the idea that any occupational activity (and even leisure activities) are related to positive health risks, the German Implementation Team decided to consider only the additional i.e. the marginal risk which is introduced by the choice of using a specific technology. The so-called net-risk takes into account only the difference between the risks of average industrial activities and the specific activity related to the fuel cycle of concern.

$$\text{Risk}_{\text{net}} = (r_{\text{activity}} - r_{\text{industrial average}}) \times \text{Work effort}_{\text{activity}}$$

where:

- \(\text{Risk}_{\text{net}}\) = net risk;
- \(r_{\text{activity}}\) = accident rate of the specific activity, normalised to a unit work effort (e.g. deaths per person-year);
- \(r_{\text{industrial average}}\) = average accident rate of all industrial activities, normalised to a unit work effort (e.g. deaths per person-year);
- \(\text{Work effort}_{\text{activity}}\) = required work effort (e.g. person-years).

The concept of net-risk has major implications on the comparison of impacts between different fuel cycles, in particular if work intensive technologies, such as photovoltaics, are considered. This method has not yet been universally applied in the ExternE Project, but is likely to receive greater use in the next phase of the study.
6.9 Conclusions

Occupational health effects can be divided into accidents and occupational diseases. Occupational accidents involve an immediate direct physical impact on the worker, with an obvious relationship between cause and effect. In contrast, occupational diseases generally occur as a more or less delayed response to a continuous, often long-term, exposure of an external burden, e.g. airborne pollutants, noise, vibration, etc.

The damage function approach is appropriate for the prediction of long-term occupational disease. For such diseases, the use of current statistics will not provide an accurate assessment of incidence, as today's disease rates will have arisen from historical exposure levels, typically at much higher concentrations. The only fuel cycle activity for which such a detailed approach has been possible to date is coalmining. A detailed methodology for predicting likely endpoints has been presented in this Chapter, though the valuation of these endpoints still requires some further work.

A detailed damage-function approach has not been possible for other activities in the fossil and renewable fuel cycles. The main reason for this is a lack of readily available data. For some fuel cycles, this is thought to significantly underestimate potential impacts, for example, there are possible long-term occupational diseases in the offshore oil and gas industry. These areas are highlighted as future research priorities.

For shorter term occupational disease and occupational accidents, calculation of occupational impacts has been based on statistics. This is regarded as a special case of an impact pathway, in which health data are linked to activities required for electricity production. Calculating occupational health impacts from statistical data implicitly assumes that the marginal impact of an additional facility is equal to the average impact. Provided that data is chosen carefully to be representative of current practice, this assumption should be reasonably valid.

A description of the data sources and accident frequencies by activity has been presented in this Chapter, based on the UK and German implementations. A summary table listing the data currently available from the ExternE Project is presented below. It is hoped that as the Project progresses a more detailed table can be built up, categorising data across the EU and from other important areas (such as non-EU coal mining activities). This can then be used as a source of reference for future studies of this nature.
Table 6.19 Summary of available occupational accident data in ExternE by fuel cycle stage.

<table>
<thead>
<tr>
<th>Fuel Cycle</th>
<th>Data Available</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Extraction</strong></td>
<td>(rate/tonne extracted)</td>
</tr>
<tr>
<td>Coal mining - accidents</td>
<td>UK, DE,</td>
</tr>
<tr>
<td>- occupational disease 1</td>
<td>UK, DE</td>
</tr>
<tr>
<td>Lignite mining</td>
<td>DE, GR</td>
</tr>
<tr>
<td>Limestone quarrying</td>
<td>DE, UK</td>
</tr>
<tr>
<td>Oil and gas extraction</td>
<td>UK, NW</td>
</tr>
<tr>
<td>Uranium mining and milling 2</td>
<td>FR</td>
</tr>
<tr>
<td>Forestry (biomass)</td>
<td>PT, GR</td>
</tr>
<tr>
<td><strong>Transport</strong></td>
<td>(rate/tonne moved or rates/km)</td>
</tr>
<tr>
<td>Rail</td>
<td>UK, DE</td>
</tr>
<tr>
<td>Barge</td>
<td>DE</td>
</tr>
<tr>
<td>Road</td>
<td>UK, DE, PT, GR, FR</td>
</tr>
<tr>
<td>Marine tanker transport</td>
<td>DE</td>
</tr>
<tr>
<td>Pipeline</td>
<td>UK, NW</td>
</tr>
<tr>
<td><strong>Processing and power plant operation</strong></td>
<td>(rate/throughput or rate/employee)</td>
</tr>
<tr>
<td>Gas treatment</td>
<td>UK, NW</td>
</tr>
<tr>
<td>Oil refining</td>
<td>DE, UK</td>
</tr>
<tr>
<td>Uranium processing, enrichment and generation 2</td>
<td>FR, NL</td>
</tr>
<tr>
<td>Power station operation</td>
<td>UK, DE, PT, GR, NW, ES, NL</td>
</tr>
<tr>
<td><strong>Construction</strong></td>
<td>(rate/value)</td>
</tr>
<tr>
<td>Construction (plant)</td>
<td>UK, DE, PT, GR, FR, NW, IT, ES, NL</td>
</tr>
<tr>
<td>Dismantling</td>
<td>2% of construction costs</td>
</tr>
<tr>
<td><strong>Other</strong></td>
<td></td>
</tr>
<tr>
<td>Manufacture of materials (renewable tech.)</td>
<td>UK, DE, GR</td>
</tr>
<tr>
<td>Transmission</td>
<td>NW, UK</td>
</tr>
</tbody>
</table>

1 quantified using the damage function approach.
2 also includes assessment of occupational radiological impacts.

Key:
DE = Germany; FR = France; GR = Greece; NL = Netherlands; IT = Italy; NW = Norway; PT = Portugal; ES = Spain; UK = United Kingdom.
6.10 References


7. IMPACTS ON TERRESTRIAL ECOSYSTEMS

7.1 Introduction

This chapter primarily addresses the assessment of the impacts of acidic deposition and ozone, produced as a consequence of energy use, on natural and semi-natural terrestrial ecosystems. Several aspects of this analysis are common to assessment of impacts to agriculture and commercial forestry. Reference to these types of ecosystem is also made, though further details are provided in Chapters 8 and 9.

Treatment of the impacts of global climate change is discussed in chapter 12. Other effects of the fuel cycles on terrestrial ecosystems result from the siting of power stations, and activities associated with (e.g.) coal mining, limestone extraction and waste disposal. These are typically less important than the effects of air pollution because they act over a restricted range. However, they may be significant if an ecosystem of special value (for whatever reason) is damaged. Consideration of such cases is too specific to particular energy projects to consider in the ExternE Project. However, methods for costing this type of damage are briefly reviewed by Markandya (European Commission, 1995).

Until recently most pollution problems were caused by large point sources such as the Trail Smelter in British Columbia near the Canadian/US border (NRCC, 1939). Damage was frequently severe but was usually limited to a relatively small area. The source of pollution could be easily identified and measures could be taken to reduce or internalise damage. However, since the early 1970s effects of pollutants associated with fossil fuel combustion have been detected on agriculture, forestry and natural ecosystems at levels previously thought to be too low to cause damage. Many of the problems observed today result from long range transport of pollutants over distances of several hundred kilometres or more.

Damage related to chronic pollutant exposure is usually different to the acute injury caused by point sources. It is often not possible to accredit such effects to a specific stress agent. Multiple stress hypotheses have now been proposed to account for forest decline and loss of natural ecosystems in a number of regions. Although these are appealing, the variety of mechanisms identified and the lack of good process oriented models makes it very difficult to assess the impact of an individual stress such as a specific pollutant.

Long range transport of pollutants presents difficulties for our analysis because it means that many impacts need to be assessed over a very large area. The pollutant increment caused by a power plant at a site many kilometres away will be very small, and associated impacts at that site, attributable to the power station, will be almost negligible. However a vast area of land is affected by emissions from any power station and damage may be significant when summed over all sites. The perceived importance of these effects is demonstrated by the fact that impacts of acidification on ecosystems have been the driving force behind agreements to reduce emissions of acidifying pollutants made under the UN ECE Convention on Long-Range Transboundary Air Pollution. Long range effects cannot therefore be ignored.
A second problem associated with the potential for a single power plant to cause damage over huge areas of land concerns the extent to which it is appropriate to transfer data and models from one country or region to another. Consideration should also be given to variation in the use of mitigating measures between different countries. Transferability of valuation procedures may be more difficult for several reasons including variation in the value systems of the inhabitants of different regions.

Data reflecting risk and distribution of ecosystems are available and have been integrated into the political process regarding the reduction of acidifying emissions in Europe. The methodology used for this work is described in this Chapter. However, there appears to have been only one attempt to use this data to value damage to ecosystems (ECOTEC, 1994). Whilst this demonstrates the possibility of an economic assessment being carried out, further work is needed to see how widely the results of the contingent valuation study carried out for the ECOTEC study can be applied.

### 7.2 Impact Pathways for Acidic Deposition and Photo-Oxidants

Impact pathways have been developed to describe the progression of effect from power station emission to valuation for crops, forests and wild plants and animals. The design has been unified so that they are comparable and easy to follow. A pathway describing effects on natural flora and fauna is shown in Figure 7.1.

All known effects, including feedbacks have been included, whether or not these are thought to be quantifiable at the current time. On the basis of a multiple-stress hypothesis it should be assumed that each effect on an organism may interact with any other impact on that organism, whether they are joined by an arrow or not (depiction of all potential interactions would be confusing). The comprehensive nature of these pathways is intended to allow the effects that have been quantified to be put into perspective with those that have not. Consideration of all potential impacts will also assist in the identification of priorities for future research.

Care is needed during the implementation of pathways. In particular it is essential that:
- All species and habitats that might be affected are considered;
- The assessment is not (as far as possible) truncated to a limited geographical area to simplify the analysis;
- Uncertainty is assessed for any estimate of damage.
Figure 7.1 An impact pathway typical of those used to describe the effects of acidic deposition and photo-oxidants on ecosystems. This example shows impacts to natural terrestrial ecosystems. All known effects have been included in this pathway, whether they are quantifiable using current knowledge or not.
7.2.1 Implementation of pathways

The following series of stages is required to implement the pathway in full:
1. Identification of relevant pollutants;
2. Identification of reference environment data such as:
   i) Existing pollutant levels/loads,
   ii) Sensitivity of ecosystems,
   iii) Distribution of species,
   iv) Existing problems (e.g. forest decline),
3. Definition of a grid system relevant to the impact under analysis, and entry of reference environment data into that system;
4. Quantification of emissions;
5. Modelling of atmospheric chemistry and transport to calculate the increment in concentration and deposition in each cell of the grid defined in (3) resulting from energy use;
6. Identification of critical loads and levels, and suitable exposure-response relationships;
7. Calculation of the impact of the pollutant concentration with and without the increment resulting from the operation of the power plant under investigation;
8. Assessment of the accuracy with which impacts may be estimated;
9. Identification of impacts that cannot yet be modelled;

7.2.2 Pollutants relevant to the pathway

The importance of the different atmospheric pollutants of principal concern to this study in terms of their potential effect on terrestrial ecosystems is summarised in Table 7.1. Freshwater fisheries are included for comparison. Although levels of peroxy-acetyl nitrate (PAN) are increasing in Europe they have not been considered, as reported levels (UK PORC, 1990) appear to be well below the threshold figure identified in a review by Taylor et al (1986).

Ammonia is released from some fuel cycles in small quantities, as a result of gasification from water in cooling towers and as a fugitive emission from selective catalytic reduction. The principal source of this gas in the atmosphere is intensive agriculture. It is probably not present at concentrations that are directly phytotoxic on anything more than a very local scale (Taylor et al, 1986), but will add to total acid and nitrogen depositions and increase the rate at which SO$_4$ is deposited, and vice versa (McLeod et al, 1990). Biological effects of ammonia deposited in this way have been recorded for a number of ecosystems (van der Eerden et al, 1990; Holland et al, 1995). It would thus be wrong to exclude NH$_3$.

For agriculture there is good agreement that in general terms the order of importance of the pollutants shown in Table 7.1 (in terms of their potential to cause damage) is O$_3$ > SO$_2$ > total acidic deposition > total N deposition, which is in agreement with the findings of NAPAP (Shriner, 1991). Some of these pollutants may actually increase crop yield, depending on other conditions, through fertilisation effects. For other ecosystems the ranking will vary. Low productivity ecosystems are especially sensitive to nitrogen inputs.
Table 7.1 The importance of major fuel cycle air pollutants in the pathways associated with acidic deposition and photo-oxidants and forests, crops, wild plants, animals in terrestrial ecosystems and freshwater fisheries. Ammonia has been included because of the importance of its interaction with SO₂.

<table>
<thead>
<tr>
<th>Pollutant</th>
<th>Forests</th>
<th>Crops</th>
<th>Natural flora</th>
<th>Natural fauna</th>
<th>Fisheries</th>
</tr>
</thead>
<tbody>
<tr>
<td>SO₂</td>
<td>XX</td>
<td>XX</td>
<td>XX</td>
<td>X</td>
<td></td>
</tr>
<tr>
<td>NOₓ</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td></td>
</tr>
<tr>
<td>NH₃</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td></td>
</tr>
<tr>
<td>O₃</td>
<td>XXX</td>
<td>XXX</td>
<td>XXX</td>
<td>X</td>
<td></td>
</tr>
<tr>
<td>Total acid</td>
<td>XXX</td>
<td>X</td>
<td>XXX</td>
<td>X</td>
<td>XXX</td>
</tr>
<tr>
<td>Total nitrogen</td>
<td>XXX</td>
<td>X</td>
<td>XXX</td>
<td>X</td>
<td>XX</td>
</tr>
</tbody>
</table>

Key:
Blank = Not ecologically significant.
X = Indirectly important as a result of interactions with other ecosystem components (see text).
XX = Identified as having significant impacts in some areas.
XXX = Highly important and believed to have direct and significant effects over potentially large areas of Europe.

Current atmospheric concentrations of gaseous pollutants in most parts of Europe will only affect animals indirectly through effects on plants used for food or shelter.

7.2.3 Characterisation of reference environments

A large amount of data is required to properly characterise the reference environment for ecological impacts. In all cases pollutant concentrations are needed, expressed in a form that is suitable for the damage function to be used. In addition the following lists show the type of information required to make a full assessment of the impacts of atmospheric pollutants. A partial assessment can, however, be made from a more limited dataset.

Forests:
1. Maps of soil characteristics and critical loads;
2. Distribution of forest species across Europe;
3. Distribution of forest species by elevation;
4. Distribution of damage classes (from national surveys);
5. Distribution of areas particularly susceptible to climatic extremes;
6. Distribution of forest pests and/or pesticide usage.

Crops:
1. Distribution of crop species;
2. Distribution of yield of each species;
3. Distribution of soil types;
4. Distribution of pest damage and/or pesticide usage;
5. Distribution of areas particularly susceptible to climatic extremes.
Critical loads maps are not necessary for assessment of effects on intensively managed agricultural land as management practices bypass many of the activities that would normally be performed by soil organisms. These practices include liming of fields to stop soil acidification which is caused by agriculture (through harvest) as well as by acidic deposition (see UK TERG, 1988).

**Natural ecosystems:**
1. Critical loads maps;
2. Distribution of ecologically important sites;
3. Distribution of species likely to be affected by acid deposition and photo-oxidants;
4. Distribution of habitats likely to be affected by acidic deposition and photo-oxidants.

The scales at which each impact is modelled should reflect both the amount of detail contained in the databases to be used and the precision of the transport models. In a few countries some data are available down to a resolution of 1 km by 1 km. This is too fine for this type of work, in terms not only of the precision of the data itself and models of atmospheric transport and chemistry, but also in terms of the amount of information that would need to be stored and manipulated. In some cases it may be reasonable to analyse an impact at 2 different scales to reflect concern over particular resources that are close to the fuel cycle activity in question.

### 7.3 Critical Loads and Levels

Critical loads and critical levels refer to the maximum exposure to a given pollutant that an ecosystem can tolerate without experiencing damage of some kind. They have been defined for several pollutants and ecosystems. However, they cannot be used directly to assess damages per se, rather they simply identify the areas in which damage is likely to occur. The critical loads approach has of course been used successfully for the determination of policy through the framework of the UNECE Convention on Long-Range Transboundary Air Pollution.

#### 7.3.1 Critical loads

The term ‘critical load’ has been defined by a number of authors. The following definition was given by UNECE (1990);

*“The highest deposition of acidifying compounds that will not cause chemical changes leading to long term harmful effects on ecosystems structure and function according to present knowledge.”*

A number of approaches have been developed for setting or calculating critical loads of acidity, sulphur and nitrogen (Federal Environment Agency, 1993):

1. The critical load is assigned on the basis of information on system properties which control the response of the system to inputs of the relevant pollutant;
2. The critical load is calculated using a simple empirical model;
3. The critical load is calculated using a simple mass balance model which balances the sources and sinks of the given pollutant in the target system;
4. The critical load is determined using a dynamic model which incorporates the main ecosystem properties and processes which control system response to the pollutant inputs.
Empirical critical loads of acidity for soils have been assigned on the basis of soil mineralogy and chemistry (Nilsson and Grennfelt 1988 and Table 7.2). Critical loads of nitrogen for a range of terrestrial ecosystems have been set on the basis of field experiments, observational data over time and transects across pollution gradients (Grennfelt and Thornelof, 1992, and Table 7.3). The steady state water chemistry model derived by Henriksen (1988) has been widely used to determine critical loads of acidity and sulphur for surface waters. Currently used versions of the mass balance models for acidity, sulphur and nitrogen are given in the Federal Environment Agency (1993) publication referred to above and in Grennfelt and Thornelof (1992). The dynamic models which have been applied to calculate critical loads include MAGIC (Cosby et al, 1985), SAFE (Warfvinge and Sverdrup, 1992) and RAINS (see Alcamo et al, 1990).

The concept of target load is a policy tool derived from critical loads. It is argued that it would be unrealistic to reduce emissions to ensure that critical loads are not exceeded anywhere in Europe because of processes of natural acidification, and the extreme sensitivity of certain sites. However, in order to formulate realistic policy objectives it is necessary to agree on a maximum load to prevent unacceptable levels of damage. Assessment of a target value for a reasonably sized area becomes extremely difficult (Chadwick and Kuylenstierna, 1990) because of the variety of variables that influence critical load. For the purposes of this study critical loads should be used to define areas subject to damage rather than target loads, the use of which would lead to underestimation of impacts.

**Table 7.2** Critical loads of acidity and sulphur for forest soils (compiled from Nilsson and Grennfelt, 1988).

<table>
<thead>
<tr>
<th>Class</th>
<th>Minerals controlling weathering</th>
<th>Usual parent rock</th>
<th>Total acidity (kmol H+ km² year⁻¹)</th>
<th>Equivalent of sulphur (kg ha⁻¹ year⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Quartz K-feldspar</td>
<td>Granite Quartzite</td>
<td>&lt; 20</td>
<td>&lt; 3</td>
</tr>
<tr>
<td>2</td>
<td>Muscovite Plagioclase Biotite (&lt;5%)</td>
<td>Granite Gneiss</td>
<td>20 - 50</td>
<td>3 - 8</td>
</tr>
<tr>
<td>3</td>
<td>Biotite Amphibole (&lt;5%)</td>
<td>Granodiorite Greywakee Schist Gabbro</td>
<td>50 - 100</td>
<td>8 - 16</td>
</tr>
<tr>
<td>4</td>
<td>Pyroxene Epidote Olivine (&lt;5%)</td>
<td>Gabbro Basalt</td>
<td>100 - 200</td>
<td>16 - 32</td>
</tr>
<tr>
<td>5</td>
<td>Carbonates</td>
<td>Limestone Marlstone</td>
<td>&gt; 200</td>
<td>&gt; 32</td>
</tr>
</tbody>
</table>
Table 7.3 Critical loads for nitrogen of different types of (semi-) natural and wetland ecosystem. Values are taken from the report of the Lokeberg workshop (Grennfelt and Thornelof, 1992). They replace the values given by Nilsson and Grennfelt (1988) which were generally lower.

<table>
<thead>
<tr>
<th>Type of ecosystem</th>
<th>Critical load (kg N ha(^{-1}) yr(^{-1}))</th>
<th>Reliability</th>
<th>Indication</th>
</tr>
</thead>
<tbody>
<tr>
<td>Acidic (managed) coniferous forest</td>
<td>15-20</td>
<td>Reasonable</td>
<td>Changes in ground flora and mycorrhizal fruit bodies</td>
</tr>
<tr>
<td>Acidic (managed) deciduous forest</td>
<td>&lt;15-20</td>
<td>Reasonable</td>
<td>Changes ground flora</td>
</tr>
<tr>
<td>Calcareous forests</td>
<td>Unknown(^1)</td>
<td>-</td>
<td>Unknown</td>
</tr>
<tr>
<td>Acidic (unmanaged) forest</td>
<td>Unknown(^1)</td>
<td>-</td>
<td>Unknown</td>
</tr>
<tr>
<td>Lowland dry-heathland</td>
<td>15-20</td>
<td>Good</td>
<td>Transition of heather to grass</td>
</tr>
<tr>
<td>Lowland wet-heathland</td>
<td>17-22</td>
<td>Good</td>
<td>Transition of heather to grass</td>
</tr>
<tr>
<td>Species-rich lowland heaths/acid grassland</td>
<td>7-20</td>
<td>Reasonable</td>
<td>Decline of sensitive species</td>
</tr>
<tr>
<td>Arctic and alpine heaths</td>
<td>5-15</td>
<td>Reasonable</td>
<td>Decline of lichens, mosses and evergreen dwarf shrubs, increase in grasses and herbs</td>
</tr>
<tr>
<td>Calcareous species-rich grassland</td>
<td>14-25</td>
<td>Good</td>
<td>Increased tall grasses, reduction of diversity</td>
</tr>
<tr>
<td>Neutral-acid species-rich grassland</td>
<td>20-30</td>
<td>Reasonable</td>
<td>Increased tall grasses, reduction of diversity</td>
</tr>
<tr>
<td>Montane-subalpine grassland</td>
<td>10-15</td>
<td>Reasonable</td>
<td>Increased tall graminoids, reduction of diversity</td>
</tr>
<tr>
<td>Shallow soft-water bodies</td>
<td>5-10</td>
<td>Good</td>
<td>Decline of isoeid species</td>
</tr>
<tr>
<td>Mesotrophic fens</td>
<td>20-35</td>
<td>Reasonable</td>
<td>Increased tall graminoids, reduction of diversity</td>
</tr>
<tr>
<td>Ombrotrophic bogs</td>
<td>5-10</td>
<td>Reasonable</td>
<td>Decline of typical mosses, increased tall graminoids</td>
</tr>
</tbody>
</table>

\(^1\) Values given by Nilsson and Grennfelt (1988) for deciduous and coniferous forests were 5-20 and 3-15 kg N ha\(^{-1}\) yr\(^{-1}\) respectively. Precise figures are dependent on the condition of the forest and its productivity. For declining or mature forests Nilsson and Grennfelt stated that the critical load would approach 0 kg N ha\(^{-1}\) yr\(^{-1}\).

7.3.2 Critical levels

UNECE (1990) give the following definition of critical levels;

‘The concentration of pollutants in the atmosphere above which direct adverse effects on receptors such as plants, ecosystems or materials may occur according to present knowledge.’
Current recommendations for critical levels are given in Table 7.4. Values have in most cases been set on the basis of empirical field observations or chamber studies.

**Table 7.4.** Critical levels of SO₂, NOₓ, O₃, and rain and cloud composition by exposure duration and land use.

<table>
<thead>
<tr>
<th>Pollutant</th>
<th>Land use category</th>
<th>Exposure duration</th>
<th>Level, µg m⁻³ unless otherwise stated</th>
</tr>
</thead>
<tbody>
<tr>
<td>NO₂¹</td>
<td>All</td>
<td>Annual mean</td>
<td>30</td>
</tr>
<tr>
<td></td>
<td></td>
<td>4 hour mean</td>
<td>95</td>
</tr>
<tr>
<td>O₃</td>
<td>All</td>
<td>Annual</td>
<td>300 ppb.hours above 40 ppb baseline</td>
</tr>
<tr>
<td>SO₂</td>
<td>Forests</td>
<td>Annual and winter mean</td>
<td>15/20²</td>
</tr>
<tr>
<td></td>
<td>Natural vegetation</td>
<td>Annual and winter mean</td>
<td>15/20²</td>
</tr>
<tr>
<td></td>
<td>Crops</td>
<td>Annual and winter mean</td>
<td>30</td>
</tr>
<tr>
<td></td>
<td>Cyanobacterial lichens</td>
<td>Annual mean</td>
<td>10</td>
</tr>
<tr>
<td>NH₃</td>
<td>All</td>
<td>1 hour</td>
<td>3300</td>
</tr>
<tr>
<td></td>
<td></td>
<td>1 day</td>
<td>270</td>
</tr>
<tr>
<td></td>
<td></td>
<td>1 month</td>
<td>23</td>
</tr>
<tr>
<td></td>
<td></td>
<td>1 year</td>
<td>8</td>
</tr>
<tr>
<td>Rain and cloud</td>
<td>Crops</td>
<td>Growing season mean</td>
<td>1 mmol H⁺ l⁻¹</td>
</tr>
<tr>
<td></td>
<td>Bryophytes/ lichens</td>
<td>1 hour (single episode)</td>
<td>3 mmol H⁺ l⁻¹</td>
</tr>
<tr>
<td></td>
<td>Forests³</td>
<td>Annual total</td>
<td>600 mol H⁺ ha⁻¹ yr⁻¹</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Time weighted annual mean⁴</td>
<td>0.3 mmol (H⁺ or NH₄⁺) l⁻¹</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>with 0.15mmol SO₄²⁻ l⁻¹</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Annual mean</td>
<td>1 µg S m⁻³ as particulate sulphate</td>
</tr>
</tbody>
</table>

¹ Critical level for NO₂ in combination with SO₂ and O₃ at concentrations below their critical level.
² The lower figure is suggested for forests where the Effective Temperature Sum > 5°C is < 1000°C
³ Undefined where Ca and Mg concentrations exceed H⁺ and NH₄⁺ concentrations.
⁴ This mean to be applied where ground-level cloud persists for >10% of the time.
7.4 Available Methodologies for Assessment of Impacts on Semi-Natural Communities

7.4.1 Methods
A number of approaches are being explored for defining the magnitude and geographical extent of the impacts of natural ecosystems and species to atmospheric pollution:

1. **Critical loads/levels** - as described above. These can be combined with information on stock at risk and modelled pollution data to define the geographical area, and proportion of the stock at risk and where damage would therefore be expected, in response to given pollutant deposition or pollutant concentration scenarios;

2. **Mechanistic growth models** - models incorporating factors which control growth and responses to pollutant inputs have been developed for a small number of vegetation communities. Notable amongst these are the CALLUNA and ERICA models developed in the Netherlands to assess the impact of enhanced nitrogen deposition, and the interactions with climatic and herbivore stress on the respective types of heathland. Models of this type are important as they allow trends over time to be examined; to date the models have not been tested outside the Netherlands;

3. **Linked hydrochemical and water chemistry-biota status models** - Ormerod *et al* (1990) linked the MAGIC hydrochemistry model with a model relating the type of aquatic invertebrate community to water chemistry. The outputs from the hydrochemical model provided inputs to an invertebrate model which then predicted the change in invertebrate community in response to changes in water chemistry. The approach parallels that developed to assess the impact of changes in water chemistry on brown trout populations (Ormerod *et al*, 1990). A similar approach could be used to assess impacts of changes in water chemistry on populations and/or breeding success of Dippers (Cinclus cinclus) and Grey Wagtails, based on the relationship between breeding success of these bird species and water chemistry (Ormerod *et al*, 1986).

7.4.2 Stock at risk
All the approaches, and any alternative methodologies, require the stock at risk to be defined and delimited. Stock at risk could be defined as, for example:

1. Specific, highly valued species or habitats, for example Red List species;
2. Species or habitats known to be sensitive to the pollutant under consideration;
3. The habitats and species within high value sites such as Nature Reserves, World Heritage Sites or Sites of Special Scientific Interest;
4. The whole stock of natural or semi-natural habitats.

Whether a species or habitat based definition is used, information is required on the distribution and size of the resource. The availability and quality of such databases varies considerably between countries. For example, the UK has databases of the occurrence of most species of the British flora on a 10 km grid cell basis, a Land Cover Map (derived from satellite imagery) which can be used to indicate the occurrence of some cover types/habitats at the level of 30 m pixels, and habitat maps for most counties of England at a scale of 1:25,000.
7.4.3 Application of a critical loads based approach: Impacts of nitrogen deposition.

Currently, the most widely applicable approach for analysis of pollutant effects on terrestrial ecosystems is the critical loads/levels approach. The application of this approach is illustrated here by an assessment of the impacts of nitrogen deposition on certain sensitive vegetation types in Great Britain. The approach requires:
1. Modelled, spatially disaggregated data on nitrogen deposition;
2. Spatial data on stock at risk;
3. Critical loads for the stock at risk.

Stock at risk

Nitrogen sensitive vegetation communities were first identified, using published information. The distribution of these was then mapped using the following methodology, applied to one of the vegetation types: montane-subalpine grassland. The plant species typical of this type of community were first identified, using published records (Rodwell et al., 1991). The number of species, from a list of 16 indicator species in this community, occurring in each 10 km grid cell in the UK were then identified and plotted using the databases of the Biological Records Centre (BRC) (Figure 7.2). The stock at risk was then defined as those 10 km squares in which more than a specified number of the species, 4 for the case of the montane-subalpine grassland, occurred. The choice of the number of species used as a cut-off when defining the stock at risk was chosen following discussions with research workers with extensive knowledge of this type of community.

Critical load

The critical load currently set for montane-subalpine grassland (Grennfelt and Thornelof, 1992, see Table 7.3) was then assigned to the 10 km squares containing the stock at risk.

Analysis of exceedance of critical load

The critical load map was then overlaid with data on oxidised nitrogen deposition, in this example mean annual deposition 1989 - 92, to identify the areas in which the critical load is exceeded by deposition of oxidised nitrogen alone (i.e. ignoring inputs of reduced nitrogen), and by how much (Figure 7.3). This can be used to determine the proportion of the stock at risk likely to be damaged by the given rates of deposition. Modelled deposition data for alternative emission - deposition scenarios can be similarly overlaid onto the critical load map to determine changes in the proportion of the stock at risk likely to be damaged.

The maps show that montane-subalpine grasslands are to be found mostly in the north-western parts of Great Britain, in upland areas. In this case only a small area (0.7% of the UK total for this community type) is likely to be affected by inputs of oxidised nitrogen. A much larger area (70% of the UK total for montane-subalpine grassland) would be affected if inputs of reduced and oxidised nitrogen were added together.
Figure 7.2 Distribution of montane-subalpine grassland in Great Britain. Characteristic species per 10 km square. Data supplied by the Biological Records Centre, ITE Monks Wood.
Figure 7.3 Exceedance of critical load for montane-subalpine grassland by oxidised nitrogen deposition. A much greater area of exceedance (covering 70% of the shaded area) would be calculated if reduced and oxidised nitrogen were considered together.
The results are presented in this way to demonstrate that the impacts are not solely the result of energy consumption. In order to achieve critical loads, emission reductions are required from other areas of human activity. Marginal analysis of the effects of emissions of NO\textsubscript{x} from a power station upon ecosystems would of course require the effects of that power station to be estimated against total (both oxidised and reduced) nitrogen deposition.

The analysis performed here is intended to show that it is possible to carry out a detailed impact assessment, in spite of the fact that analysis of impacts to natural ecosystems in economic terms has not yet been possible within the ExternE Project. Analysis has been disaggregated with respect to geographical area, community type and pollutant. Analysis on this basis can clearly provide an important input to the development of policy for environmental protection.

7.5 Summary

It is widely accepted that terrestrial ecosystems are at risk from deposition of atmospheric pollutants released through the combustion of fossil fuels. The extent of this risk is dependent on the type of ecosystem concerned and the level of other stresses, whether of natural or anthropogenic origin. These include climatic factors, pest levels and possibly unforeseen effects of management techniques. This Chapter has concentrated on assessment of natural terrestrial ecosystems, though some of the information provided is also applicable to agriculture and forestry, which are discussed in further detail in the next two chapters.

It is not currently possible to quantify damage to natural ecosystems in economic terms at the scale required for analysis of the impacts of air pollutants produced by the main fossil fuel cycles for electricity generation. We have demonstrated that it is possible to identify the species and habitats that are at risk from exposure to the pollutants for which critical loads and levels have been defined. Such information has been used recently as the basis for contingent valuation work in the UK (ECOTEC, 1994). However, more such studies, in a variety of countries, are necessary before costs can be assessed at a pan-European level with any confidence. It may be appropriate to direct research more towards the use of tools for multi-criteria analysis, which would allow comparison of results expressed in a variety of units, than to wait for the availability of sufficient valuation data to allow meaningful analysis.
7.6 References


NRCC (1939). Effect of Sulphur Dioxide on Vegetation, National Research Council of Canada, Ottawa, Canada.


8. IMPACTS OF ACIDIC DEPOSITION AND PHOTO-OXIDANTS ON AGRICULTURE

8.1 Effects of Pollutants on Agriculture

8.1.1 The impact pathway

The impact pathway developed to show the sequence of effects from power station emission to valuation of economic damage of agricultural systems is shown in Figure 8.1. The pathway concentrates on impacts on crops, rather than impacts on livestock, though, as will be seen, livestock account for roughly half of European agricultural production. The reason for this is that farm animals seem unlikely to be affected directly to any significant degree by the effects of low levels of air pollution. However, productivity may be influenced by changes in the yield of crops grown for feeding farm animals.

There are 2 basic pathways through which pollutants act on plants. The first is through foliar uptake of pollutants, and the second through effects of acidic deposition on the soil. The sequence of events was summarised by Guderian et al (1985);
1. Pollutant uptake,
2. Perturbation of cell functions and structures,
3. Attempts to re-establish normal metabolic functioning through repair and/or compensatory mechanisms (homeostasis),
4. Impact.

The manner in which cell functions and structures will be affected is pollutant dependent. In general the most sensitive metabolic pathways in plants appear to be those associated with photosynthesis.

8.1.2 The pollutants of concern

A large number of atmospheric pollutants have been found to influence crop yield or quality. Of interest to assessment of the effects of fossil fuel combustion are O₃, SO₂, NOₓ, acidic deposition, PAN, HF, HCl and heavy metals. This Chapter concentrates on the first four of these. The others are likely to be of less importance in Europe at the regional scale which is of most interest to the ExternE Project, though this needs to be kept under review. The possibility of localised problems arising in parts of Europe following emission of HF, HCl and heavy metals is not disputed though seems unlikely to make a significant contribution to the total externality of the fuel cycles under analysis.
Figure 8.1 Impact pathway illustrating the effects of air pollution on agricultural crops.
NAPAP reported that the order of the major pollutants associated with fossil fuel combustion in terms of negative impacts to crop production in the US was $O_3 > SO_2 >$ acidic deposition $> NO_x$ (Shriner, 1991). There is dispute in the literature as to whether any of these pollutants apart from $O_3$ does serious damage to crops at present levels in western Europe and the USA. The report by UK TERG (1988) concluded that;

‘Major agricultural crops in the UK are unlikely to be damaged directly by current rural concentrations of $SO_2$ and $NO_2$. Recent evidence suggests that interactions between pollutant stresses and other stresses such as pests, may be extremely important in influencing crop yields.’

A similar conclusion was reached by Shriner (1991) for North America, in the review of results from NAPAP;

‘Based on the crop-effects research conducted by NAPAP and other research programs, acidic precipitation at ambient levels in the United States has not been shown to be responsible for regional crop yield reduction.’

Van der Eerden et al (1988) estimated that air pollution reduced total crop volume in the Netherlands by 5%, 3.4% of which was caused by $O_3$, 1.2% by $SO_2$ and 0.4% by HF. They further estimated that consumers would benefit by $320$ million annually if pollutant concentrations were reduced to background levels. This study did not include any estimation of interactions between different pollutants, climate or pests, which would almost certainly have increased estimated losses further (see UK TERG, 1988). Although this is in agreement with $O_3$ being the most serious pollutant with respect to crop production, it suggests annual damage of over $100$ million in the Netherlands from other pollutants. It should be noted, however, that the exposure-response relationships used by van der Eerden may have overestimated damages, particularly for $SO_2$ (see below).

Overall, nitrogenous pollutants are regarded as being relatively unimportant for agriculture, though they may interact with $SO_2$ and $O_3$ (Elkley et al, 1988), and some other stressors, such as insect pests. There is also the potential for crops will benefit to some degree from the deposition of oxidised nitrogen which is, after all, commonly applied by farmers as a fertiliser.

It is thought that total acid deposition is unlikely to cause serious damage to agricultural soils in most areas of Europe as a result of traditional liming practices. However, the effectiveness of liming treatments is influenced partly by atmospheric inputs. Concern has been expressed in Europe recently because of falling application rates of lime in some areas (see Section 8.4, below).

### 8.1.3 Damage mechanisms

Foliar uptake of $SO_2$ may have several effects, including:

1. Loss of stomatal control (Mansfield and Freer-Smith, 1984);
2. Alterations to pH of cell fluids (Jager and Klein, 1980);
3. Chlorophyll destruction (Malhotra, 1977);
4. Alterations to the amino acid spectrum (Arndt, 1970; Ziegler, 1975);

Comprehensive overviews are given by Koziol and Whatley (1984) and Malhotra and Khan (1984). The rate of photosynthesis is affected quickly by many of these effects and hence is frequently used as a measure of plant response to pollution (see, for example, Darrall, 1986). A number of secondary effects have been reported, such as reduced tolerance to drought or cold, and increased pest performance.

The principal effects of O$_3$ are on membranes, chiefly through the peroxidation of the double bonds of unsaturated lipids. Damage to membrane-bound pumps affects the ability of cells to maintain homeostatic control. Reaction with components of membranes also releases free radicals that will interfere with many other cell processes. O$_3$ is too reactive to enter and accumulate within cells.

Nitrogenous compounds are rarely found at atmospheric concentrations considered to be phytotoxic. At typical atmospheric concentrations plants can generally metabolise NO$_x$ and NH$_3$ at a rate sufficient to avoid accumulation. Although direct effects of exposure to current ambient levels of nitrogenous pollutants seem unlikely, secondary effects such as reduced tolerance of other environmental stresses are potentially serious.

Exposure of crops at high altitude to acid mists is of limited importance for agriculture at the European level, though may have significant effects in some areas.

In any assessment of externalities most emphasis is laid upon detrimental impacts. However, it is important to note that crops in some areas have been found to grow better in the presence of low levels of fossil fuel related pollution than without (Roberts, 1984; Clarke and Murray, 1990; Murray and Wilson, 1990; McLeod et al., 1991). The reason for this is simple; sulphur and nitrogen are essential nutrients for living organisms. A number of results also show an apparent stimulation of growth caused by low levels of O$_3$ (Adaros et al, 1991b; Skärby et al, 1994). Skärby et al postulated that this effect could arise through plants being adapted to present day O$_3$ levels, and that deviation from the norm is detrimental to plant performance. An alternative explanation offered was that plants exposed to non-filtered air (the ‘polluted’ treatment, in contrast to the control using charcoal filtered air) would benefit from a higher exposure to sulphur and nitrogen pollutants, which at low levels may have a fertilisational role, depending on the availability of these nutrients from the soil. However, Skärby et al regarded this mechanism as insufficient to fully explain the observed response.

8.1.4 Variability of response

Most experimental research in this area has investigated the direct effects of pollutants on crops. Relatively little attention has been given to interactions between pollutants and other stresses on agricultural crops. A given dose of a pollutant will produce a variable response depending on a wide range of factors, including:

1. Age of organism/tissue (Shaw et al, 1993);
2. Other pollutants (Mansfield and McCune, 1988; Jäger and Schulze, 1988);
3. Time of day or season (Baker and Fullwood, 1986; Baker et al., 1986);
4. Temperature (Mansfield et al., 1986);
5. Water status and relative humidity (Mansfield et al., 1986);
6. Light conditions (Mansfield et al., 1986);
7. Soil and plant nutrient status (Schulze et al., 1989);
8. Species/cultivar (Taylor et al., 1986);
9. Interactions with pests and pathogens (Warrington, 1989; Houlden et al., 1990);
10. Pollution climate.

Some effects are transient such as the growth response of plants to chronic fumigation treatments. Growth of over-wintering cereal crops tends to be reduced by SO\textsubscript{2} during winter but recovers as the weather improves (Colvill et al., 1983; Pande and Mansfield, 1985; Baker et al., 1986). The most likely explanation for this behaviour is that plants growing slowly (in the winter) are not capable of detoxifying SO\textsubscript{2} as quickly as they take it up, whilst plants growing faster in the spring and summer can and may benefit by using it as a fertiliser.

Jager and Schulze (1988) concluded that the effects of combinations of pollutants are mainly additive or synergistic in the range of concentrations that are typical of Western Europe. They surmised that these responses would be most evident at or near the threshold concentrations for any given pollutants. In contrast Adaros et al. (1991a; 1991b), studying the effects of O\textsubscript{3}, SO\textsubscript{2} and NO\textsubscript{x} on barley, wheat and rape over several seasons, found that most interactions between pollutants were antagonistic. Unfortunately only a few experiments have been performed using mixtures of pollutants at anything like realistic concentrations. A fully integrated assessment of the effect of changes in the overall pollution climate by a reference power plant on, for example, crop yield, is thus not possible.

Factors such as water availability and temperature vary greatly from year to year. Some information is available on the manner in which they affect plant response to pollution, though this is probably insufficient to allow estimation of the resulting damage. The interaction between O\textsubscript{3} and water stress was specifically investigated during the NCLAN program, though the results are not definitive (Somerville et al., 1989; Heck, 1989).

Numerous studies have been conducted on interactions between pollutants and aphids, producing remarkably similar results - pests proliferate most on plant material that has been exposed to pollutants. The principal reason for this is that pollutants improve the nutritional value of plants through effects on amino acid composition (Bolsinger and Flukiger, 1989; Riemer and Whittaker, 1989). Suitable exposure-response data for use in the ExternE Project have not yet been identified for assessment of this interaction. The situation is the same for interactions between pollutants and pathogens.

A major problem with much of the published data is that the pollutant levels used are not relevant to conditions that prevail today. Concentrations of any of the pollutants of concern here in excess of 100 ppb are unusual across most of Europe. Annual mean levels are very much less than this. This is compounded by the fact that the sensitivity of plants to acute pollutant exposures (as summarised by Taylor et al., 1986) is a poor guide to the sensitivity of plants to chronic exposures of the same pollutants, further limiting the use of many studies for
Impacts of Air Pollution on Agriculture

our purposes (see below). Ayazloo and Bell (1981) and Horsman et al (1979), each studying the responses of different genotypes of a single species of grass, and Garsed and Rutter (1982) investigating the response of different species of tree, found no relationship between the ranking of sensitivity to chronic exposures and that for acute fumigation. Different mechanisms may be responsible for determining sensitivity to the 2 types of dose.

Some of the most interesting studies in recent years are those in which the effects of growing plants in ambient air are compared with the effects of growing plants in filtered air, from which a proportion of the ambient pollution has been removed (e.g. Temmerman et al, 1992; Vandermeiren et al, 1992). Although the results of this work may be less dramatic than those achieved using acute pollutant doses, they do show that current ambient pollution levels are capable of affecting crops.

Two other interactions which may be important were reported from an open-air fumigation in the UK. The results of this experiment are particularly important because of the fact that plants were grown under field conditions. The first was an interaction between spray damage and SO\(_2\) (Baker and Fullwood, 1986). The second was an interaction between SO\(_2\) and lodging (collapse of crops before harvest) (Baker et al, 1990) which can cause significant crop losses. A possible mechanism for this was identified by Pande and Mansfield (1985) who observed that fumigated plants grew taller than non-fumigated, presumably in response to the fertilisational effects of this pollutant. Tall plants are known to be more susceptible to lodging than shorter ones (Austin, 1980). This may be a case where the fertilising effect of a pollutant has a direct negative effect on harvested yield. Extrapolation of these results is, however, unwarranted at the present time given that the observations were restricted to a single experiment.

Although we have not identified functions that can be used to model these interactions explicitly, some of them may be modelled implicitly, depending on the design of the experiments that provide exposure-response data. For this reason data from experiments which have exposed crops in the field is to be preferred to data from exposure under more confined conditions. This issue is further addressed below.

Economic damage associated with particular impacts is also dependent on the species in question and the way in which that species is valued. For example, foliar appearance of crops affects saleability of some species (e.g. spinach, lettuce and cut flowers), but not others (e.g. onions, wheat and rhubarb) unless injury is severe enough to reduce yield.

8.1.5 Experimental approaches to study plant response to pollution

It is not surprising that extensive variation is frequently observed between the results of different experiments, given the number of factors that may influence plant response to pollution. Some understanding of the advantages and disadvantages of the approaches used is important when evaluating the results of pollution research, particularly for older studies. The following techniques have been used to assess the response of plants to pollution. There is an inevitable trade-off between price, the realism of growing conditions and the amount of control that can be exerted over the experimental system. The best experimental method in any case is dependent on the hypothesis under examination.
i) Controlled environment facilities.
These range in size from small chambers with volumes less than 1 m$^3$ to rooms with floor spaces of 10 m$^2$ or more. Controlled environment chambers are only suitable for small specimens and give the user only limited choice over conditions. They have probably been used to best effect for biochemical analyses. In the most advanced controlled environment rooms ("Ecotrons") light, temperature, relative humidity, day length and soil parameters can be controlled as well as air quality. Although they sound ideal for this type of work they are expensive to build and maintain and hence only a few are available. There are thus limits on the amount of replication possible when using these systems.

ii) Open/semi-open top chambers.
These chambers are the most widely used system for the study of pollutant effects on plants. They allow air quality to be controlled accurately either by the addition of pollutants, or by filtration of air prior to its passage through the chamber. Growing conditions, especially with reference to microclimate (and soils if plants are grown in pots) differ to those of plants growing outside chambers, although designs have been refined over the years to reduce ‘chamber effects’ (Unsworth, 1991). Improvements have also been made in the design of chamber fumigation/ventilation systems, making the manner in which plants are exposed to pollutants much more realistic than in the past. Plant size is strictly limited by chamber size.

iii) Field exposure systems.
Elevated pollutant concentrations may be achieved using a network of pipes designed to give even exposure horizontally across a large area of a crop or forest and vertically through the canopy (McLeod et al., 1985). These systems are generally more expensive than open top chambers as they use greater quantities of pollutants and require more sophisticated control gear to alter pollutant levels. These systems have the advantage over chamber systems in that growing conditions are closer to normal. This has to be balanced against the fact that control of pollutants is more difficult, and, at times (high or low wind speeds or extreme pollutant events) impossible. Large scale field exposure systems may be used by several groups of researchers at the same time, which can give important insights into the mechanisms controlling certain types of interaction.

It should be noted that Skärby et al. found significant differences in response to ozone between plants grown in pots and plants grown in the field.

iv) Pollution gradient exposures.
A number of experiments and surveys have been conducted using plants sited at known distances from discrete pollutant sources such as the Trail smelter (NRCC, 1939) or from cities such as London (Ashmore and Dalpra, 1985). Conditions in this type of experiment can be almost totally natural. There is, however, no control over conditions, including pollutant exposure, beyond that exerted by deciding where to place experimental units.
8.2 European Crop Production

8.2.1 Total production figures

Table 8.1 provides a listing of agricultural products in order of their total value in the European Community in 1989, using data from EUROSTAT (1990). The overall total is split evenly between the production of animals/animal products, and crops, though the top three categories are all associated with animals (milk, cattle and pigs).

As noted above, in Section 8.1.1, there is no evidence to suggest that livestock will be affected directly by air pollution at levels typical of those in rural areas of western Europe. Any impacts on farm animals are thus likely to be associated with changes in the availability of their foodstuff. For animals that graze in the open, such as most cattle and sheep, it is thus reasonable to assess effects through analysis of impacts on pasture grass. Analysis of impacts on species that are typically reared more intensively (such as pigs and poultry) may not be necessary, as a reasonable approximation may be possible through assessment of changes in the production of crops grown for animal feed.

Variation in the ranking between different countries is illustrated in the Table by reference to Germany, Greece, Ireland, Spain and the UK. Variation arises largely because of climate, with clear north/south differences in the production of grapes, olives, rice, tobacco and citrus fruits.

Prominent among the categories of crop production shown in Table 8.1 are ‘fresh vegetables’, ‘other crops’, ‘fresh fruit’ and ‘oil seeds and fruit [excluding olives]’. Together they account for 21.7% of European agricultural production. Each of these categories contains many species, for example, ‘fresh vegetables’ contains 64 categories, ‘fresh fruit’ has 27. Although individually each of the species covered in these categories may make only a minute contribution to European agricultural output, the total is significant. A substantial degree of generalisation is therefore necessary for a comprehensive analysis, as exposure-response data are not available for many of the species grown. At the present time we have not attempted to provide a comprehensive analysis, instead concentrating on those crops for which reasonable data are available. Expansion of the analysis will need to be considered as part of the next phase of the ExternE Project. However, at the present time the data shown in Table 8.1 provide information which can be used to assess the extent to which our analysis underestimates total impacts on agriculture.

Some work has been carried out to assess the sensitivity of a range of crops to a variety of pollutants (Taylor et al, 1986), in terms of the pollutant levels required to cause visible injury to plant tissues (which may or may not affect yield or saleability of produce). Unfortunately most of the studies referred to by Taylor were conducted at very high concentrations, which do not necessarily provide much guidance on the sensitivity of plants at lower, more realistic concentrations (see Section 8.1.4, above). Extreme examples are exposures of white clover to 100,000 ppb NH₃ for 0.1 hours and potato to 4200 ppb SO₂ for 1 hour. Barley, for example, is described as being ‘sensitive’ to O₃, whereas major studies in both Europe and the USA have
failed to show any response. Oilseed rape is described as ‘moderately sensitive’ to SO$_2$, when experimental data show that it grows better when exposed to levels that are significantly higher than those currently prevailing in rural areas of Europe. Sensitivity data, such as that used by Taylor et al must therefore be regarded with a strong degree of scepticism, and have not been used within the present study.

Table 8.1 Agricultural production in the European Community in 1989 (EUROSTAT, 1990). Products are ranked by their total value in million ECU and % of total EC agricultural production. For comparison, the rankings for 5 individual countries are shown.

<table>
<thead>
<tr>
<th>EU</th>
<th>MECU</th>
<th>%</th>
<th>Germany</th>
<th>Greece</th>
<th>Ireland</th>
<th>Spain</th>
<th>UK</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Milk</td>
<td>34,824</td>
<td>17.3</td>
<td>1</td>
<td>4</td>
<td>2</td>
<td>3</td>
</tr>
<tr>
<td>2</td>
<td>Cattle</td>
<td>25,407</td>
<td>12.7</td>
<td>3</td>
<td>9</td>
<td>1</td>
<td>5</td>
</tr>
<tr>
<td>3</td>
<td>Pigs</td>
<td>21,657</td>
<td>10.8</td>
<td>2</td>
<td>10</td>
<td>4</td>
<td>2</td>
</tr>
<tr>
<td>4</td>
<td>Fresh vegetables</td>
<td>17,500</td>
<td>8.7</td>
<td>11</td>
<td>2</td>
<td>7</td>
<td>1</td>
</tr>
<tr>
<td>5</td>
<td>Wheat and spelt</td>
<td>12,543</td>
<td>6.3</td>
<td>6</td>
<td>7</td>
<td>10</td>
<td>10</td>
</tr>
<tr>
<td>6</td>
<td>Other crops ¹</td>
<td>12,124</td>
<td>6.1</td>
<td>4</td>
<td>11</td>
<td>9</td>
<td>7</td>
</tr>
<tr>
<td>7</td>
<td>Grape must and wine</td>
<td>11,700</td>
<td>5.8</td>
<td>6</td>
<td>16</td>
<td>ng</td>
<td>8</td>
</tr>
<tr>
<td>8</td>
<td>Fresh fruit</td>
<td>9,211</td>
<td>4.6</td>
<td>5</td>
<td>6</td>
<td>14</td>
<td>4</td>
</tr>
<tr>
<td>9</td>
<td>Poultry</td>
<td>8,633</td>
<td>4.3</td>
<td>13</td>
<td>15</td>
<td>6</td>
<td>9</td>
</tr>
<tr>
<td>10</td>
<td>Other animals ¹</td>
<td>6,351</td>
<td>3.2</td>
<td>16</td>
<td>1</td>
<td>3</td>
<td>6</td>
</tr>
<tr>
<td>11</td>
<td>Eggs</td>
<td>5,204</td>
<td>2.6</td>
<td>9</td>
<td>14</td>
<td>12</td>
<td>13</td>
</tr>
<tr>
<td>12</td>
<td>Barley</td>
<td>4,694</td>
<td>2.3</td>
<td>10</td>
<td>24</td>
<td>5</td>
<td>12</td>
</tr>
<tr>
<td>13</td>
<td>Sugar beet</td>
<td>4,686</td>
<td>2.3</td>
<td>8</td>
<td>19</td>
<td>8</td>
<td>17</td>
</tr>
<tr>
<td>14</td>
<td>Oil seeds and fruit ²</td>
<td>4,504</td>
<td>2.3</td>
<td>12</td>
<td>25</td>
<td>ng</td>
<td>16</td>
</tr>
<tr>
<td>15</td>
<td>Potatoes</td>
<td>4,401</td>
<td>2.2</td>
<td>14</td>
<td>17</td>
<td>11</td>
<td>15</td>
</tr>
<tr>
<td>16</td>
<td>Maize</td>
<td>3,898</td>
<td>1.9</td>
<td>19</td>
<td>12</td>
<td>ng</td>
<td>18</td>
</tr>
<tr>
<td>17</td>
<td>Olive oil</td>
<td>2,987</td>
<td>1.5</td>
<td>ng</td>
<td>3</td>
<td>ng</td>
<td>14</td>
</tr>
<tr>
<td>18</td>
<td>Citrus fruit</td>
<td>2,382</td>
<td>1.2</td>
<td>ng</td>
<td>18</td>
<td>ng</td>
<td>11</td>
</tr>
<tr>
<td>19</td>
<td>Pulses</td>
<td>1,436</td>
<td>0.7</td>
<td>20</td>
<td>21</td>
<td>17</td>
<td>23</td>
</tr>
<tr>
<td>20</td>
<td>Tobacco</td>
<td>1,236</td>
<td>0.6</td>
<td>21</td>
<td>8</td>
<td>ng</td>
<td>22</td>
</tr>
<tr>
<td>21</td>
<td>Other industrial crops ¹</td>
<td>1,165</td>
<td>0.6</td>
<td>ng</td>
<td>5</td>
<td>16</td>
<td>20</td>
</tr>
<tr>
<td>22</td>
<td>Other animal products ¹</td>
<td>939</td>
<td>0.5</td>
<td>17</td>
<td>22</td>
<td>13</td>
<td>19</td>
</tr>
<tr>
<td>23</td>
<td>Grape</td>
<td>922</td>
<td>0.5</td>
<td>ng</td>
<td>13</td>
<td>ng</td>
<td>21</td>
</tr>
<tr>
<td>24</td>
<td>Other cereals ¹</td>
<td>826</td>
<td>0.4</td>
<td>15</td>
<td>26</td>
<td>15</td>
<td>26</td>
</tr>
<tr>
<td>25</td>
<td>Rice</td>
<td>723</td>
<td>0.4</td>
<td>ng</td>
<td>23</td>
<td>ng</td>
<td>25</td>
</tr>
<tr>
<td>26</td>
<td>Table olives</td>
<td>350</td>
<td>0.2</td>
<td>ng</td>
<td>20</td>
<td>ng</td>
<td>24</td>
</tr>
<tr>
<td>27</td>
<td>Hops</td>
<td>147</td>
<td>0.1</td>
<td>18</td>
<td>ng</td>
<td>ng</td>
<td>27</td>
</tr>
</tbody>
</table>

¹ Relates to products not otherwise listed in the table.
² Excluding olives, data for which are given separately in the table.

‘ng’ denotes crops for individual countries that are not grown in sufficient quantity to register in the EUROSTAT summary tables.

8.2.2 Sources of reference data for describing crop distributions

Originally the analysis of impacts of fuel cycle emissions on agriculture within the ExternE Project was conducted only for crops grown within the country that was host to the fuel cycle activity under assessment. This allowed the use of detailed databases in Germany and the UK, as follows;
Impacts of Air Pollution on Agriculture

Germany

Source for the local range assessment; Statistisches Landesamt für Baden-Württemberg (Statistical Office Baden-Württemberg, StaLA). StaLA gathers these data by two surveys on the agricultural sector:
1. Bodennutzungserhebung (BO) (Survey on land use);
2. Harvest Statistics.

Source for the German national assessment: Statistische Bundesamt (German Statistical Office, StaBA, 1990). A distinction between winter and spring wheat and barley was made because of differences in yield (of between 10 - 30%) and SO₂ exposure. The areas on which wheat and barley are grown are so large that this distinction is meaningful. This is not the case for any of the other crops considered so far in the project.

UK

Source for UK local and national assessment; ITE Land Classification and Database. This database provides data on the distribution of crops by area sown. To translate these figures into yield, as required for the exposure-response functions given below, mean yield data were taken from MAFF (1990) and averaged over the years 1984 to 1988.

Analysis at the European level

Subsequent analysis using models of atmospheric transport and chemistry revealed the potential for these impacts to operate at the pan-European level, and so a central database was established within the EcoSense Model developed by IER, using data taken from the EUROGRID database. This covers the whole of the European Union. It was originally developed by the Institut de Protection et de Sûreté Nucléaire (ISPN) for the European Commission’s Radiation Protection Programme. Some of the information needed was not available from EUROGRID. The database was thus supplemented using the Länderberichte published by StaBA.

Future analysis within the ExternE Project may require additional data to be input to the database held within EcoSense, either to improve the quality of existing numbers or to extend the analysis to additional crops. This data should distribution of crops by yield. However, it is likely that in many countries the only geographically disaggregated data available will describe the areas under each crop. These can be converted to yield using estimates of national mean yield per unit area, which should either be published directly or can be calculated from figures showing the total yield, and area under, each crop. This approach was adopted for the UK assessment, as described above. It is desirable to use figures averaged over a number of years to reflect the annual variability of crop production.
8.3 Exposure-Response Functions

There has been considerable debate as to whether plant response to pollution is dictated primarily by extreme events or by long term mean levels of exposure. In particular cases this is likely to be dependent on the type of damage observed. However, there is currently general consensus that yield changes are more closely related to long term mean levels of pollution than to peak values (e.g. Garsed and Rutter, 1982; Baker et al, 1986).

A problem common to almost all experiments is that effect on yield is estimated by comparison with plants that are themselves exposed to pollutants, albeit at lower concentrations. For the derivation of damage functions the assumption of no effect of pollution on plants in the least polluted treatment is almost certainly wrong. However, it is unavoidable unless a vast amount of data are available.

8.3.1 Sulphur dioxide

The species for which there is most exposure-response data for direct effects of SO$_2$ is perennial ryegrass (*Lolium perenne* L.). This is the species most commonly used in pastures in Europe, and hence is important because of the contribution of livestock grazing on pastures to total European crop production (see Section 8.2). Research on this species began when natural populations were observed to be tolerant of ambient pollution levels in NW England whereas newly introduced varieties which should have grown faster became chlorotic and grew poorly in response to local SO$_2$ levels (Bell and Mudd, 1976).

Van der Eerden *et al* (1988) used a relationship originally derived (OECD, 1980) from results for *L. perenne* to estimate yield losses of all crops in the Netherlands;

\[
\% \text{ Yield Loss} = s \frac{100 \ e(x)}{1 + e(x)}
\]

where:

\(s\) = sensitivity index (estimated for each crop)

\(x = 3.8 \log (C - 5.8) - 9.2\)

\(C = \text{annual 7 hr (10.00 to 17.00) daily mean SO}_2\ \text{concentration, } \mu g \ m^{-3}\).

Derivation of the sensitivity index for each crop species was rather arbitrary (by the admission of the authors). Rather than assessing simply the effect of pollution on yield, the index sought to reflect impacts on economic value of different crops, including, for example, sensitivity of price to foliar injury of crops such as spinach. Values are shown in Table 8.2. Most of the sensitivity values are less than one, implying that equation 1, when used independently of any sensitivity factor, is likely to overestimate damages. Interestingly van der Eerden *et al* use a factor of only 0.3 for grasses, though the relationship was developed from work on a species of grass.
Table 8.2  Sensitivity index developed by van der Eerden et al (1988).

<table>
<thead>
<tr>
<th>Species</th>
<th>Index</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cereals</td>
<td>0.6 - 0.8</td>
</tr>
<tr>
<td>Legumes</td>
<td>0 - 1.0</td>
</tr>
<tr>
<td>Grass</td>
<td>0.3</td>
</tr>
<tr>
<td>Potato</td>
<td>0.3</td>
</tr>
<tr>
<td>Cabbage</td>
<td>0.3</td>
</tr>
<tr>
<td>Cucumber</td>
<td>0.6</td>
</tr>
<tr>
<td>Lettuce</td>
<td>3</td>
</tr>
<tr>
<td>Spinach</td>
<td>0.9</td>
</tr>
<tr>
<td>Fruit</td>
<td>0.6 - 1.0</td>
</tr>
<tr>
<td>Floriculture</td>
<td>0 - 1.8</td>
</tr>
<tr>
<td>Arboriculture</td>
<td>0.3</td>
</tr>
</tbody>
</table>

Equation 1 was re-assessed by Roberts (1984) against all available data. He analysed 125 long time-exposure chamber studies described in the literature. The results for 21 different crop species fumigated with SO$_2$ concentrations between 16 and 263 ppb under controlled conditions were used to calculate 16 functions. The contribution of the following factors to the variability of data was investigated:

1. Differences in response between plant species;
2. Differences in pollutant flux (see Ashenden and Mansfield, 1977);
3. Variation related to growth rate and time of year;
4. Variation with plant age;
5. Variation with plant density.

From the equations given by Roberts the following have been selected, in which $y$ = % yield loss and SO$_2$ is expressed as mean concentration (ppb):

$$y = -3.18 - 0.188\text{SO}_2$$  \hspace{1cm} (2)

$r^2 = 0.18$, $p < 0.001$, 115 data points covering 21 species.

exposure time $> 20$ d

SO$_2$ concentration $> 15$ ppb (40 µg m$^{-3}$), $< 200$ ppb (572 µg m$^{-3}$)

$$y = 6.4 - 0.31\text{SO}_2$$  \hspace{1cm} (3)

$r^2 = 0.12$, $p < 0.05$, 35 data points, 9 species.

exposure time $> 20$ d

SO$_2$ concentration $> 17$ ppb (45 µg m$^{-3}$), $< 50$ ppb (143 µg m$^{-3}$)

$$y = 2.75 - 0.18\text{SO}_2$$  \hspace{1cm} (4)
Impacts of Air Pollution on Agriculture

\[ r^2 = 0.35, p < 0.001, 45 \text{ data points for } L. \text{ perenne only.} \]

exposure time > 20 d,  
\[ \text{SO}_2 \text{ concentration } > 15 \text{ ppb (40 mg m}^{-3} \text{)}, < 200 \text{ ppb (572 mg m}^{-3} \text{)} \]

\[ y = 7.33 - 0.21(\text{SO}_2) \]  \hspace{1em} (5)

\[ r^2 = 0.66, p < 0.001, 33 \text{ data points for } L. \text{ perenne only.} \]

exposure time > 20 d, more than one air change/minute,  
\[ \text{SO}_2 \text{ concentration } > 18 \text{ ppb (48 mg m}^{-3} \text{)}, < 200 \text{ ppb (572 mg m}^{-3} \text{)} \]

Unlike OECD (1980) Roberts found the best fit for linear functions. Comparing relationships for \textit{L. perenne} from the 2 studies, Figure 8.2 clearly shows that equation 4 accounts for much more of the variation in response than equation 1. Equation 1 has thus been rejected from our analysis.

![](image)

**Figure 8.2** Comparison of exposure-response functions from Roberts (1984) and OECD (1980), plotted against available data for \textit{L. perenne}.

Baker \textit{et al} (1986) produce the following function from work on winter barley;

\[ \% \text{ Yield Loss} = 9.35 - 0.69(\text{SO}_2) \]  \hspace{1em} (6)

Where \text{SO}_2 = \text{annual mean SO}_2 \text{ concentration, ppb.}
This relationship was derived from the results of field fumigation of winter barley (cv. Igri) over 2 seasons (1983/4 and 1984/5) and is shown in Figure 8.3 with the original data. One problem with equations 2 to 6 is that none of them extend below an SO_2 concentration of about 15 ppb, corresponding to 0% yield reduction. As it has been demonstrated in a large number of experiments that low levels of SO_2 are capable of stimulating growth it cannot be assumed that there is no effect on yield below 15 ppb, nor can it be assumed that any effect will be detrimental. As few rural locations in Europe experience SO_2 levels greater than 15 ppb, these equations are not directly applicable. 2 approaches were taken to resolve this, the first being to estimate the worst case, whereby these equations were applied directly over the full range of SO_2 levels. The second approach was to estimate a curve that fitted the following criteria, to produce an exposure-response of the form suggested by Fowler et al (1988):

1. 0% yield reduction at 0 ppb and value predicted by the equation;  
2. Maximum yield increase at an SO_2 concentration midway between the 2 values for which 0% yield effect is predicted from (1);  
3. The experimentally predicted line formed a tangent to this curve.

The parameters of quadratic functions were estimated using the regression facility within Lotus-123 to give the best fit to the above criteria. The curves were applied for SO_2 concentrations between the 2 that predicted yield effect = 0%. The experimentally derived relationships were used at higher concentrations. Taking equations 4 and 6 this approach gave the following sets of exposure-response functions, in which the concentration of SO_2 is expressed in ppb and \( y = \% \) yield loss;

**Baker 1:**  
\[ y = -0.69(SO_2) + 9.35 \]  

**Baker 2:**  
\[ y = 0.74(SO_2) - 0.055(SO_2)^2 \]  
\[ y = -0.69(SO_2) + 9.35 \]  
\[ \text{(above 13.6 ppb)} \]  

**Roberts 1:**  
\[ y = -0.18(SO_2) + 2.75 \]  

**Roberts 2:**  
\[ y = 0.20(SO_2) - 0.013(SO_2)^2 \]  
\[ y = -0.18(SO_2) + 2.75 \]  
\[ \text{(above 15.3 ppb)} \]

An example showing the extrapolation procedure is shown in Figure 8.3.

It is important to ascertain precisely what stresses may be included in any exposure-response relationship. Equations 2 to 5 used results from a large number of experiments conducted in chambers of varying designs. Given the extensive scatter in the data (Figure 8.2) it seems likely that effects of a range of stresses have been averaged out through regression. However, interactions between pollutants and pests and pathogens are probably not accounted for because control measures may be applied more stringently during experiments in chambers than in the field. Prolonged drought, exposure to low temperatures and effects of increased soil acidity would also not be accounted for very well because of experimental design and the duration of experiments.
Impacts of Air Pollution on Agriculture

Figure 8.3 Extrapolation of exposure-response functions below the lowest exposure level used experimentally.

Baker et al (1986) reported that weather conditions varied greatly between years in their experiment, although there was a high degree of consistency in their results;

‘1983/4 had an ordinarily cold winter and a dry, sunny summer, but the winter of 1984/5 was severe in January and February and the summer was dull and wet’.

Mean O₃ and NOₓ concentrations were around 19 ppb and 24 ppb, respectively. The soil was a sandy loam. Management practices in this work reflected those typical of local farms, fertiliser and agrochemicals being applied at the same times and rates. No records of pest or pathogen performance are given in the paper.

Weigel et al (1990) studied several crop cultivars common in Germany. Two spring barley cultivars (‘Arena’, ‘Hockey’), two bean cultivars (‘Rintintin’, ‘Rosisty’) and one rape cultivar (‘Callypso’) were exposed to five different SO₂ levels between 7 and 202 µg m⁻³ (2.5 - 70 ppb) in open-top chambers. Exposure periods ranged from 49 to 96 days. 8 h/daily mean O₃-concentration ranged between 14 and 19 µg m⁻³. Daily means of NO₂ and NO concentrations were generally lower than 10 µg m⁻³. Yield increases appeared in all SO₂ treatments for the rape cultivar compared with controls whereas beans and barley were quite SO₂ sensitive. The probable cause of the positive response of rape was the high sulphur demand of this species.
Data for barley and beans were taken from this paper and used to calculate the following relationships ($SO_2$ in $\mu g \ m^{-3}$):

$$y = 4.92 - 0.26(SO_2)$$

\(r^2 = 0.73, p < 0.001, 20\) data points for barley and beans
\((y = 4.92 - 0.74(SO_2), SO_2 \text{ in ppb})\)

$$y = 10.92 - 0.31(SO_2)$$

\(r^2 = 0.73, p < 0.01, 10\) data points for barley only
\((y = 10.92 - 0.89(SO_2), SO_2 \text{ in ppb})\)

$$y = -0.93 - 0.21(SO_2)$$

\(r^2 = 0.84, p < 0.001, 10\) data points for beans only
\((y = -0.93 - 0.60(SO_2), SO_2 \text{ in ppb})\)

The background mean $SO_2$ concentrations that provided the control levels in this study were low ($7 - 9 \ \mu g \ m^{-3}$, about 3 ppb). It is considered that the functions derived from this data, unlike those given by Roberts (1984) and Baker et al (1986), may thus be applied directly without the need to consider how best to extrapolate back to 0 ppb $SO_2$. Therefore a negative effect of increased $SO_2$ on growth is anticipated by these functions at all concentrations above zero.

This raises an important issue; are we correct in not extrapolating the Baker and Roberts functions back to zero by a simple linear approach? In spite of the results of Weigel for barley and beans, we believe that the quadratic extrapolation is appropriate. Indeed, other data from Weigel for oilseed rape (yet to be used in this study) showed a stimulation of growth at all applied $SO_2$ levels. The precise form of the function (and hence the point at which stimulation gives way to depression of growth) is dependent upon the conditions under which crops are grown. A possible reason for the observed negative response to all concentrations is that the barley and bean plants studied by Weigel may have been grown in a soil with high levels of sulphur. This is not universally the case in the agricultural areas of Europe. This issue is identified as the most important uncertainty affecting our analysis of the effects of $SO_2$ on crop growth.

8.3.2 Ozone

Exposure-response functions from North American studies

The use of the Weibull function (Rawlings and Cure, 1985) to describe exposure-response relationships for $O_3$ on crops is now well accepted (e.g. Atkinson et al, 1985) following the results of the extensive NCLAN study in the USA (Somerville et al, 1989). This function takes the following form (see also Figure 8.4).
$y_r = a \cdot e^{-(x/s)c}$  \hspace{1cm} (12)

where

$y_r =$ crop yield,

$a =$ hypothetical yield at 0 ppm ozone, usually normalised to 1,

$x =$ a measure of ozone concentration,

$s =$ ozone concentration when yield = 0.37,

$c =$ dimensionless exponential loss function to reflect sensitivity.

Concentrations of O$_3$ are expressed in terms of a 7 hr (09:00 to 16:00) or 12 hr (07.00 to 19.00) daily mean. Values of the parameters $s$ and $c$ are available for several crop species and are shown in Table 8.3. However, some of these data appear to have been used by Fuhrer (1989) to derive Weibull function parameters (see Table 8.3). There is thus an outstanding issue of which approach should be recommended where duplicate functions exist.

![Figure 8.4](image-url)

**Figure 8.4** Illustration of the form of the Weibull exposure-response curve for three crops of varying sensitivity to O$_3$ assessed in the US NCLAN study.

For grapevines, Fuhrer et al (1989) also derived the following relationship,

$$Y_{rel} = 1.121 - 6.63 \cdot x_{12}$$  \hspace{1cm} (13)

where $x_{12} =$ seasonal 12 hr/day average O$_3$ level.

**Table 8.3** Weibull function parameters for different crop species based on studies carried out under the NCLAN programme. $s =$ ozone concentration when yield = 0.37, $c =$ dimensionless...
Impacts of Air Pollution on Agriculture

exponential loss function to reflect sensitivity. The relevant ozone exposure metric is in ppb expressed as the seasonal 7 or 12 hour/day mean, as shown (to convert for use against ozone measured in µg m\(^{-3}\), the parameter \(s\) can be multiplied by 2, assuming temperature = 293 K, pressure = 101.3 kPa). All functions shown were derived using US data. Figures in parentheses denote estimated approximate standard errors.

<table>
<thead>
<tr>
<th>Crop</th>
<th>O(_3) metric</th>
<th>(s)</th>
<th>(c)</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Alfalfa</td>
<td>12 hr/day</td>
<td>178 (2.8)</td>
<td>2.07 (0.55)</td>
<td>Somerville et al, 1989</td>
</tr>
<tr>
<td>Barley</td>
<td>no response</td>
<td></td>
<td></td>
<td>Somerville et al, 1989</td>
</tr>
<tr>
<td>Corn (Zea mays)</td>
<td>12 hr/day</td>
<td>124 (0.2)</td>
<td>2.83 (0.23)</td>
<td>Somerville et al, 1989</td>
</tr>
<tr>
<td>Cotton</td>
<td>12 hr/day</td>
<td>111 (0.5)</td>
<td>2.06 (0.33)</td>
<td>Somerville et al, 1989</td>
</tr>
<tr>
<td>Forage grass</td>
<td>12 hr/day</td>
<td>139 (1.5)</td>
<td>1.95 (0.56)</td>
<td>Somerville et al, 1989</td>
</tr>
<tr>
<td>Kidney bean</td>
<td>7 hr/day</td>
<td>279 (7.9)</td>
<td>1.35 (0.70)</td>
<td>Somerville et al, 1989</td>
</tr>
<tr>
<td>Soybean</td>
<td>12 hr/day</td>
<td>107 (0.3)</td>
<td>1.58 (0.16)</td>
<td>Somerville et al, 1989</td>
</tr>
<tr>
<td>Wheat</td>
<td>7 hr/day</td>
<td>136 (0.6)</td>
<td>2.56 (0.41)</td>
<td>Somerville et al, 1989</td>
</tr>
<tr>
<td>Sugar beet, turnip*</td>
<td>7 hr/day</td>
<td>94</td>
<td>2.905</td>
<td>Fuhrer et al, 1989</td>
</tr>
<tr>
<td>Spinach*</td>
<td>7 hr/day</td>
<td>135</td>
<td>2.08</td>
<td>Fuhrer et al, 1989</td>
</tr>
<tr>
<td>Lettuce*</td>
<td>7 hr/day</td>
<td>122</td>
<td>8.837</td>
<td>Fuhrer et al, 1989</td>
</tr>
<tr>
<td>Tomato*</td>
<td>7 hr/day</td>
<td>142</td>
<td>2.369</td>
<td>Fuhrer et al, 1989</td>
</tr>
</tbody>
</table>

Further data exists from the NCLAN study which may be useful in the derivation of functions for lettuce, peanut, sorghum and tobacco (Somerville et al, 1989).

**Exposure-response functions from the European Open Top Chamber Programme**

The following objections have been raised relating to the use of NCLAN derived relationships within Europe:

1. The climate in northern Europe in particular is very different to that in parts of the US such as California where much of the NCLAN work was performed. Plants growing under very dry conditions will conserve water by restricting stomatal conductance, thereby also decreasing the rate of foliar \(O_3\) uptake. This appears to account for the observation that plants exposed to levels of \(O_3\) as high as 300 ppb in California appear to be suffering little or no ill effect (Heath, 1992);

2. NCLAN treatments were designed to give constant daily \(O_3\) exposures. High levels of \(O_3\) are an episodic phenomenon in Europe, and most agricultural regions experience levels greatly in excess of about 30 ppb on only a few occasions each year (Unsworth, 1991);

3. The pollution climate in Europe is more complex than in the USA. Significant levels of \(NO_x\), \(NH_3\) and \(SO_2\) in the atmosphere may influence plant response to \(O_3\) (Unsworth, 1991);

4. Some major European crops were not used in the NCLAN study. Of those that were, results may not be applicable in Europe because of differences between the sensitivity of cultivars grown in the Europe and those studied in the US.

Exposure-response functions describing the action of ozone on crops have recently been developed using European data (Skärby et al, 1994). However, only 3 crops were covered,
Impacts of Air Pollution on Agriculture

spring wheat, oats and barley, though the last 2 of these were found to be insensitive to O₃. The expert panel on crop damage convened under the ExternE Project concluded that rye was also unlikely to be sensitive to O₃. They also concluded that, in the absence of other European data, the function for spring wheat could be applied also to winter wheat, and should be used in preference to the NCLAN functions for the above reasons;

\[ Y_{rel} = 1 + 0.0008 \cdot x_{8} - 0.000075 \cdot x_{8}^2 \quad (14) \]

It should be noted that in a similar series of experiments in Europe, the pod weight of beans was only markedly affected by seasonal 8 hr/day mean O₃ levels above 140 µg m⁻³ (70 ppb).

8.3.3 Application of exposure-response functions

Yield losses from the exposure-response functions given above are expressed as a % of yield in the absence of the pollutant of interest (Y₀). However, crops in the field are clearly exposed to some level of pollution and hence the data used to define Y₀ already incorporates some pollutant effect (Figure 8.10). Direct application of the dose response functions given above will thus introduce some degree of error. To account for this, impacts were estimated by multiplying the yield at prevailing pollutant concentrations by the ratio;

\[ \frac{\% \text{ loss at incremental pollutant levels}}{\% \text{ loss at prevailing pollutant levels}} \]

Taking, for example, equations 6 and 7(a, b). Y₀ for these equations does not correspond to the original 100% figure to which they were calculated, as the published relationship has been extrapolated directly back to 0 ppb. The functions were thus used in the following forms, where;

SO₂ = existing pollutant concentration (ppb)
SO₂* = ambient pollutant concentration + increment from fuel cycle (ppb)

Equation 6:

\[ \frac{100 + 9.35 - 0.69[SO₂*]}{100 + 9.35 - 0.69[SO₂]} - 1 \right) \cdot Y \]

Equations 7a, for SO₂ < 13.6 ppb:

\[ \frac{100 + 0.74[SO₂*] - 0.055[SO₂]^2}{100 + 0.74[SO₂] - 0.055[SO₂]^2} - 1 \right) \cdot Y \]

Equation 7b, for SO₂ > 13.6 ppb:

\[ \frac{100 + 9.35 - 0.69[SO₂*]}{100 + 9.35 - 0.69[SO₂]} - 1 \right) \cdot Y \]
8.3.4 Precise definition of impacts assessed

Having defined the exposure-response functions used in the study it is necessary to consider to what extent we have covered the impact pathway. Only direct effects of SO₂ and O₃ on a limited range of crops are included. These are limited to the yield responses of plants to pollutants taken up through the stomata. The analysis does not explicitly include:
1. Interactions with pests;
2. Interactions with pathogens;
3. Interactions with climate;
4. Effects of, and interactions with, other pollutants;
5. Yield losses on crops other than those specifically covered by exposure-response functions.

Analysis of soil mediated effects is considered below.

The reason for not including the above list of factors in this paper lies simply in a lack of data: it is not intended to reflect upon their importance. There is a body of opinion that certain of these interactions will cause much greater damage than the direct impacts on yield discussed here. It is intended to analyse a number of these effects at a later date, providing that suitable data, models and exposure-response relationships are available. It is possible that these factors are implicitly included (to varying degrees) in the dose-response functions cited, given that all plants are subject to a range of stresses. It is perhaps necessary to take a view on the degree to which the experimental exposure systems could be considered typical of agricultural conditions and practice. This in itself raises difficulties because of obvious variation in different regions of the area (Europe) of interest to this study.

8.4 Assessment of Impacts of Acidification on Agricultural Soils

8.4.1 Introduction

Soil acidification is seen as one of the major current threats to soils in northern Europe. It is a process which occurs naturally at rates which depend on the type of vegetation, soil parent material, and climate. Human activities can accelerate the rate of soil acidification, by a variety of means, such as the planting of certain tree species, the use of fertilisers, and by the draining of soils. However, the major concern in Europe is the acceleration of soil acidification caused by inputs of oxides of sulphur and nitrogen produced by the burning of fossil fuels.

UK TERG (1988) concluded that the threat of acid deposition to soils of managed agricultural systems should be minimal, since management practices (liming) counteract acidification and often override many functions normally performed by soil organisms. They suggested that the only agricultural systems in the UK that are currently under threat from soil acidification are semi-natural grasslands used for grazing, especially in upland areas. Particular concern has been expressed since the 1970’s when traditional liming practices were cut back or ceased altogether, even in some sensitive areas, following the withdrawal of government subsidies.
Concern has also been expressed in other countries. Agricultural liming applications decreased by about 40% in Sweden between 1982 and 1988 (Swedish EPA, 1990). Although liming may eliminate the possibility of soil degradation by acidic deposition in well-managed land, the efficacy of applied lime may be reduced, and application rates may need to be increased.

8.4.2 Methodology

The basis of the method is to calculate:
- The total amount of acidifying pollutant deposited to the land surface in a given area;
- The amount which falls on soils which require lime (excluding, for example, urban areas, water and soils on calcareous drifts);
- The cost of neutralising this amount of acidic deposition with lime;
- The increased acidic deposition in this area resulting from the operation of a fuel cycle;
- The additional cost of neutralising the increase in inputs to soils which require lime.

Deposition values for acidity are typically expressed in terms of kilo-equivalents (keq) or mega-equivalents (Meq). One equivalent is the weight of a substance which combines with, or releases, one gram (one equivalent) of hydrogen. When sulphuric acid is neutralised by lime (calcium carbonate):

\[
H_2SO_4 + CaCO_3 → CaSO_4 + H_2O + CO_2
\]

The following conversion factors were used in our analysis;
1 keq/ha = 10^4 keq/10 km square, or 10 Meq/10 km square.
Also, 1 Meq/100 km square = 1 eq/ha.

Each grid square within the study area is given a weighting (between 0 and 1) for the proportion of the square that would need to be limed (excluding urban areas, water and soils on calcareous drifts). So far, analysis has only been conducted for the UK, and the source of data at this point was the Soil Survey of England and Wales 1:1,000,000 Soil Map of England and Wales. The total acidifying pollution input multiplied by the soil weighting factor gave the total acidifying pollution input per unit area on soils which require lime.

100 kg CaCO₃ is sufficient to neutralise 2 kg H⁺. Accordingly the total acidifying pollution input on soils which require lime was multiplied by 50 to give the amount of lime which required to neutralise it. As the total acidifying pollution input was in eq, the amount of lime is in Mg (tonnes). One tonne per 100 km sq is equivalent to 1g/ha on average. Hence, the amount of lime in t/100 km sq is divided by 1000 to give the mean kg/ha. The price of lime is ECU 16.8/tonne.

8.5 Assessment of Impacts of Nitrogen Deposition on Agricultural Soils

Nitrogen is an essential plant nutrient, applied by farmers in large quantity to their crops. The deposition of oxidised nitrogen to agricultural soils is thus beneficial (assuming that the
dosage of any fertiliser applied by a farmer is not excessive). The analysis is conducted in the same way as assessment of effects of acidic deposition. The benefit is calculated directly from the cost of nitrate fertiliser, ECU 430/tonne of nitrogen (note: not per tonne of nitrate) (Nix, 1990). Given that additional inputs will still be needed under current conditions to meet crop N requirements there is a negligible saving in the time required for fertiliser application (if any).

8.6 Pollutant x Pest Interactions

Numerous reports are available that show that major insect pests perform better on plants exposed to pollutants than those not (e.g. Riemer and Whittaker, 1989). The mechanism for this appears to be that the host plant chemistry is altered to produce greater levels of certain amino acids that limit growth of these pests (Bolsinger and Flukiger, 1989). In spite of the increasing literature on this subject, exposure-response functions for assessment of yield loss are not available.

It may be possible to make a crude estimate of effects based on analysis of expenditure on pesticides and estimates of agricultural losses caused by pests. However, this has not been attempted yet within the ExternE Project.

8.7 Conclusions

1. Pollutants have a wide range of effects on crops. We have presented details of the methodology for assessment of the following:
   - Direct impacts of SO$_2$ on yield;
   - Direct impacts of O$_3$ on yield;
   - Increased liming requirements to compensate for acid deposition;
   - Reduced nitrogenous fertiliser requirements through deposition of oxidised N.

2. Analysis covered only a small number of the crop species of interest. Future methodological research should concentrate on increasing the number of crops that can be assessed. A second priority is to include effects of pollution mediated through interaction with insect pests.

3. Published exposure-response relationships need to be critically assessed in the light of all available data before use.

4. Sensitivity of plants to acute pollutant doses is a poor guide to their reaction to chronic exposure.
5. Depending on the location of the reference power plant it seems possible that positive or negative results may be found for the incremental contribution to both SO$_2$ and O$_3$ levels. In the case of SO$_2$ this is a consequence of the possibility of a fertilisational effect at low dose. In the case of O$_3$, it is a consequence of the action of NO$_x$ as a sink or precursor for O$_3$, depending on the ratio of NO/NO$_2$ at any point downwind of the polluting activity.

6. Although estimated effects on agriculture are small according to current ExternE estimates, they may become significant once the analysis becomes more comprehensive, including more crops and possibly interactions with other stressors. Expansion of the analysis of agricultural impacts should thus be given some priority.
8.8 References


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9. IMPACTS OF ACID RAIN AND OZONE ON FORESTS

9.1 Introduction

Trees and forests perform a wide range of useful functions. In addition to timber production these include use for recreation, protection from avalanches in mountainous areas, CO₂ uptake and storage, water management and wildlife habitat. They are in many ways fundamental to European culture, whether growing in forests or urban areas.

The problem of forest decline, and its possible association with atmospheric pollution has been the subject of much research in recent years. Declines are characterised by a given set of symptoms, which often affect trees of only one species over a restricted geographical range. Declines have long term consequences on yield or even the existence of a forest or species. It must be said that air pollution is only one of a number of stressors that affect forests. Others include climate, pests, pathogens and the consequences of unsustainable management practices.

At least 18 major declines were reported in Europe and North America between 1900 and the late 1970s. It could therefore be said that they are simply to be expected, and that we must learn to live with the problem. However, the fact that damage has been identified simultaneously in so many parts of the northern hemisphere makes the present declines particularly notable.

In Europe unusual symptoms were first noticed on silver fir in Germany during the 1970s, followed by Norway spruce, Scots pine, beech and other species. The widespread nature of the problem, in terms of area and species affected, suggested that the causal factor operated at a regional level. The only agents that seemed to meet this condition were pollution and climate stress. Pollution was further implicated in the observed declines by the fact that population density of Scots pine in Germany and the UK had earlier been shown to be correlated with SO₂ levels (Knabe, 1970; Farrar et al., 1977). SO₂ at annual mean concentrations of between 15 and 40 ppb is also known to be directly responsible for the current death of spruce within the ‘Black Triangle’ of Czechoslovakia, Poland and eastern Germany (Moldan and Schnoor, 1992).

Threshold concentrations of pollutants for direct foliar damage have conventionally been estimated from short term fumigations of seedlings at high concentration in enclosed chambers. Taylor et al (1986) gave a value for Scots pine of 300 ppb SO₂ applied for 2 hours. However, Shaw et al (1993) recently established a much lower threshold for direct SO₂ damage to needles of highly sensitive individuals in a genetically variable population of Scots pine, finding classic SO₂ induced needle tip necrosis resulting from a mean of <10 ppb over a
Impacts of Acid Rain and Ozone on Forests

period of about 10 days at the time of budbreak. This level is exceeded in many areas. This figure was obtained from observations made over 3 years in a large scale continuous open air fumigation experiment, rather than in a more conventional, but less realistic, chamber-based study. Apparently identical symptoms were observed on a number of trees exposed to ambient air in an experiment run by the UK Forestry Commission in the English midlands (D. Durrant, personal communication). The precise cause of injury in this case is uncertain, however, as pollution monitoring data was not available for the period prior to and when damage appeared.

Sensitivity to short term pollutant exposure at ambient levels appears to be genetically determined (Oleksyn, 1988; Shaw et al., 1993). This is likely to lead to the loss of highly sensitive individuals, either naturally or at thinning. Given that past levels of SO$_2$ in many areas were much higher than they are now, it seems possible that existing forests may not be affected greatly by direct action of this pollutant, as sensitive individuals will have been selected out already. The establishment and performance of new forest, however, may be affected, particularly as the most productive provenances appear to be the most sensitive (Oleksyn, 1988; Holland et al., 1995).

NO$_x$ at current levels is not believed to be capable of causing damage on its own, but may increase damage by interaction with other pollutants or other stresses. Much concern has focused on O$_3$, levels of which have risen in recent years, largely as a consequence of the expansion of motor traffic. In areas subject to high insolation this pollutant is quite capable of damaging plants, as experience from the US clearly shows. O$_3$ levels in the UK frequently exceed those recommended by the World Health Organisation and the UN ECE (Bower et al., 1991). O$_3$ levels also tend to be highest in upland areas where much land is used for forestry.

Although damage to leaves may be caused directly by exposure to pollutants, less direct effects, mediated through the soil, are probably more important. A recent study of data published since 1931 in the eastern US demonstrated that sensitive sites were being damaged by acidic deposition 20 or 40 years ago (Shortle and Bondietti, 1992). The work of Tamm (1988) in Sweden has proved that soil acidification is a serious problem even in areas of Europe that are far removed from industrial activity. Soil acidification is known to disrupt nutrient cycling within forests by increasing leaching of essential nutrients such as Ca and Mg. Increased Al mobilisation may exacerbate this problem by interfering antagonistically in the uptake of nutrient base cations. Very high concentrations of Al in the soil solution may damage roots.

Potential pollutant action on trees is summarised in the impact pathway given in Figure 9.1, which also includes interactions between pollutants and pests (Sierpinsky, 1984), climate (Cape et al., 1988) and other factors. Problems may be compounded by poor silvicultural practice. The pathway is idealised in that it seeks to show what one would seek to assess if sufficient data were available. It does not show the extent of the analysis that has been attempted. For clarity a number of interactions and feedbacks have been omitted.
Impacts of Acid Rain and Ozone on Forests

Figure 9.1 Impact pathway for effects of acidic deposition and photo-oxidants on forests.
9.2 Pollution Effects on Forest Health and Recent Declines

9.2.1 Forest health in Europe

Following the introduction of standardised surveying techniques, all countries in Europe have reported damage to some degree in terms of loss of foliage (assessed in terms of crown density) and foliar discoloration, the criteria used internationally to assess forest health. Damage has been recorded on more than 30 species of tree, deciduous and coniferous. This has been caused by herbivores, insects, fungi, abiotic agents (chiefly climate), direct action of man (including poor silvicultural practice in commercial forests), fire and pollutants.

Leaf loss >25% has been reported on more than 20% of trees in Great Britain, Denmark and Liechtenstein (leaf loss <25% is generally not thought to be detrimental to tree growth). In some areas of many countries a high proportion of trees has been killed. Overall, by the late 1980s data from the UN ECE surveys showed that about 11% of trees in Europe (excluding the USSR) had lost >25% of their leaves.

Whilst pollution is undoubtedly to blame for much of the damage reported in some countries (e.g. the Czech Republic and Poland), the case in others such as the UK is less clear. In general, observed symptoms do not match those observed following experimental fumigation. In the UK, the Forestry Commission have found that 4 of the 5 most widely planted trees have denser crowns in areas with higher levels of most forms of pollution (Innes and Boswell, 1989). This distribution may be caused by enhancement of plant nitrogen levels (through deposition of NO₃ or NH₃) and/or by variation in climate.

A number of specific decline types have been recorded, particularly for Norway spruce, the most important species in Central European timber production. These are summarised below and discussed in greater detail by Huettl (1989) and Blank et al (1988).

Type 1: Yellowing in high elevation stands. Recorded in the Fichtel and Harz mountains and Black Forest in Germany and also in Austria, France, Belgium and the Netherlands. Also seen at low elevation on acid, base-poor soils. Characterised as tip yellowing of older needles exposed to sunlight in the mid and lower crown. Leads to premature needle fall. The direct cause of this symptom is magnesium (Mg) deficiency. Trees may die, but usually only when symptoms are accompanied by frost, drought, disease etc. Symptoms may be reversed if conditions improve. This type is confined to inland areas of Europe. Elsewhere, atmospheric deposition of marine derived Mg is sufficient to prevent deficiency.

Type 2: Crown thinning at medium elevations (400-600 m). Recorded in the central chain of mountains in Germany, like type 1. Primarily affects the dominant trees in a stand. Needle loss is not always preceded by yellowing, and occurs from the inside of the canopy out, and from the base of shoots to the tip. All parts of the canopy may be affected. Often associated with low foliar nutritional status of calcium (Ca), Mg, potassium (K) or phosphorus (P). This damage type has been connected with elevated levels of SO₂ and acidic deposition.
**Type 3: Needle necrosis in older stands in southern Germany.** Typically takes the form of reddening of older needles in the lower, shaded, part of the canopy. As a result, the effect on growth may be negligible even when there is marked loss of needles. The needle cast fungi *Rhizosphaera kalkhoffii* and *Lophodermium picea* are thought to cause this decline type, in association with low foliar K concentrations, which have previously been shown to make trees less resistant to fungal attack. A similar phenomenon was reported in the last century, though not on the scale observed in recent years. Curiously, these fungi have coexisted with trees in the area for many years without causing severe damage. Trees all over southern Germany have been affected on a range of soil types, usually in areas where pollution levels are low. Rehfueess (1985) states that this epidemic may have been initiated by climatic factors such as frost shock or persistent high rainfall and humidity.

**Type 4: Chlorosis in high elevation stands in the calcareous Alps.** Affected trees are found in the Bavarian Alps and similar parts of France, Switzerland and Austria. Trees in these areas have always suffered from nutrient deficiencies, particularly concerning K, Mn and Fe in a few cases, though deficiencies rarely appear in combination. However, there was a rapid increase in yellowing in 1981 associated with reduced tree vitality and even the death of trees of all age classes. It seems likely that this has resulted from an interaction between site conditions and unfavourable weather conditions. This decline type is particularly serious in Bavaria as regeneration of trees is difficult because of an over-population of game. Dead trees leave gaps in the canopy. These enhance soil erosion, exacerbating the difficulties associated with tree establishment. Erosion is regarded as a particularly serious problem given the instability of the forest ecosystem in the region, and its role in avalanche protection.

**Type 5: Crown thinning in coastal areas.** Older spruces in the coastal plains of Germany, Belgium and the Netherlands are affected by this problem which is believed to be associated with NH₃ deposition. Reduced growth has been evident since the mid-1960s. The 1976 drought increased the problem, and growth has been reduced by between 40 and 60%. Symptoms may be associated with a deficiency of Mg and K. Although NH₃ is not emitted from fuel cycles in large quantities, there is a strong interaction between SO₂ and NH₃ which greatly enhances the deposition of both in the aqueous phase (McLeod et al, 1990).

Numerous declines of silver fir have been reported before, but never on the scale observed since the early 1970s. Since then damage has been reported from Austria, Switzerland and France (in the Vosges) as well as southern Germany. The distribution of particular symptoms among silver fir does not seem to be as discrete as the decline types observed for Norway spruce. The following symptoms have been recorded:

1. **Needle yellowing.** Affects older needles on trees of all ages, especially younger ones. Similar in a number of ways (e.g. site parameters) to type 1 Norway spruce decline.

2. **Needle necroses.** Also affects trees of all ages. Generally preceded by acute yellowing. All year classes may be affected.
3. **Crown thinning.** Arises from loss of green, yellow or necrotic needles, starting with older shaded needles. Current year shoots are lost shortly before tree death.

4. **Stork nest development.** This ‘symptom’ develops naturally in older trees as apical dominance is lost. Identification of it as a decline symptom in younger trees is controversial.

5. **Development of secondary branches.** This symptom has been observed before, but occurs now with greater frequency among declining firs with pronounced needle loss. It can lead to a reduction in timber quality. It is usually regarded as a secondary stress symptom.

Sierpinsky (1984) has related infestation of the insect pest *Dreyfusia nordmannianae* to pollution. Damage to the fine rooting system of declining firs has been noted and may be related to fungal infestation.

To these declines must be added the problems of the Black Triangle, an area covering parts of the Czech Republic, Poland and eastern Germany. In this region many trees have died because of exposure to very high levels of acidic deposition.

Among broadleaved trees most concern has focused on beech, the commonest deciduous species in central Europe. The range of symptoms observed includes crown thinning and discoloration, branch dieback, root damage and growth reduction. Severe damage has been reported in the Harz mountains and Black Forest in Germany, the Vosges mountains of France and various regions of Switzerland. A number of studies have found that foliar concentrations of nutrient cations are related to observed levels of damage.

The regular shape of most conifers, with a single main stem and regular production of shorter, more or less horizontal side branches simplifies assessment of tree health. Growth of beech (and most other deciduous species) is not nearly so deterministic. Accordingly there is wide variation in crown morphology, and the definition of what may be regarded as damage. A system has been devised that assesses tree vitality in terms of past performance so that it is possible to separate long term trends from short term influences (Roloff, 1985). However, results still need to be interpreted carefully in order to identify the cause of short term effects.

### 9.2.2 Forest health in the USA

The NAPAP study concluded that the vast majority of forests in the US are not affected by decline (Barnard and Lucier, 1991), though a number of declines have been observed in the United States in recent years;

1. **Ponderosa and Jeffrey pines in San Bernadino and other parts of California.** Symptoms, first observed in the early 1950s, took the form of chlorotic mottle and general chlorosis of older needles leading to premature leaf fall. These symptoms are now regarded as classic O₃ injury. Reduced growth and increased mortality rates have also been reported. Synergistic interactions between bark beetles and root fungi have been reported. Forest structure appears to be changing, with increased invasion of incense cedar and white fir. O₃ damage has also been reported in other areas of the western US including the Sierra
Nevada and San Gabriel mountains. These declines are discussed in more detail by Miller (1989). Damage caused by peroxy acetyl nitrate (PAN) has also been observed, but is probably irrelevant for studies in Europe (Temple and Taylor, 1983).

2. **White pines in the eastern US.** Eastern white pine, growing in the southern Appalachians at altitudes <1200 m was found to be extremely sensitive to O₃ in the early 1960s (see Bruck, 1989).

3. **Red spruce and Fraser fir in the eastern US.** Mixed red spruce and Fraser fir forest occupies areas of the Appalachians at altitudes in excess of 1400 m, having been left behind after the retreat of the glaciers 10,000 years ago. These species are favoured by high moisture levels, and lack of competition from other species resulting from the low temperatures experienced at high elevation. Fraser fir has been declining in the southern Appalachians since the 1950s, following the introduction of the balsam woolly adelgid (Dull *et al*., 1990). Acidic deposition is now known to alter the resistance of red spruce to winter injury, the primary inciting factor of its decline in the northern Appalachians over the past 30 years (Cape *et al*., 1988; Adams and Eagar, 1990). Some authors (e.g. Bruck, 1989) believe that the spruce-fir ecosystem was stressed and pre-disposed to decline even before pollutant levels in the area became an issue.

4. **Yellow pines in the south-eastern US.** Growth rates of these trees in natural stands, but not plantation, has fallen in the past 30 years. Mortality rates have also increased. It is believed that this is largely a consequence of historical land use patterns, increased stand age and competition and other natural factors. Air pollution may play a role in this decline, but this has not yet been demonstrated (Barnard and Lucier, 1991).

5. **Sugar maple in north-east America.** This is regarded as particularly serious in Quebec and parts of Ontario, Vermont and Massachusetts. Important factors related to this decline are K deficiency, defoliation by insects and climate. Air pollution is not believed to be a major causal agent (McIlveen *et al*., 1986), though it may make a subtle contribution in a few areas. In general, this decline seems to have had little effect on overall growth of sugar maple.

### 9.2.3 Forest health in other parts of the world

Forest health problems in which air pollution may be implicated have been recorded in other parts of the world. Many of the problems in the less developed nations stem from localised point sources of what are regarded as the classic air pollutants - SO₂ and HF. Yu *et al* (1990) reported just such damage in the Nanshan forest in south west China. In these cases, like the ponderosa and Jeffrey pines of San Bernadino which suffer O₃ damage, symptoms are often unequivocal and can be safely ascribed to pollution of one sort or another.

### 9.2.4 Summary of the role of pollution in forest damage

Air pollution has played a major role in several recent forest declines, though not all. However, the precise manner of pollutant action varies from site to site and species to species.
Impacts of Acid Rain and Ozone on Forests

Several common pollutants associated with the coal fuel cycle are known to be capable of causing direct damage to plants. However, this is rarely observed in the field, except in areas where SO$_2$ or O$_3$ levels are very high.

More typically the primary impact of acidic deposition is mediated through the soil. It is well accepted that this can lead to increased acidity of poorly buffered soils, increased leaching rates of base cations and increased mobile concentrations of potentially phytotoxic metal ions such as Al$^{3+}$. The evidence that aluminium damages roots directly is rather limited, though it is known to be capable of antagonistically reducing the uptake of Mg and Ca. Pollution may also increase the severity of secondary stresses: interactions between SO$_2$, NO$_x$, NH$_3$, and O$_3$, and insect pests, pathogens, cold stress and drought have all been recorded.

In some of the pollution induced declines discussed there are clear connections between forest condition and productivity (e.g. crown thinning of Norway spruce in coastal areas of Germany, Belgium and the Netherlands, O$_3$ damage to pines in San Bernadino, etc.).

However, in a number of other cases a clear link between crown condition or pollutant loading and growth has not been found. In the UK forest health survey of 1988 the condition of 4 of the 5 most widely planted species was best in the most polluted parts of the country. Elsewhere in Europe, Kenk and Fischer (1988) and Falkengren-Gerup and Eriksson (1990) have observed increased forest growth, apparently in response to increased N inputs to systems low in this macronutrient (N deposition in parts of Sweden, for example, has increased from 1-2 kg/ha/yr to 20-30 kg/ha/yr in the last 60 years: see Sverdrup et al., 1993). Although this appears to be beneficial, it is possible that the increased growth seen now, combined with continuing acid inputs to forest soils, will lead to deficiencies of other elements in the future in a similar manner to that described by Schulze et al (1989).

Effects of acidic deposition on forests appear to be much worse in Europe than in North America. Damage in many parts of Europe has been linked to acid rain (albeit often tentatively), whilst NAPAP only identified such a problem for red spruce. Elsewhere in North America deposition is regarded as too low to cause damage, at least in the short term according to NAPAP. As both continents are heavily industrialised and home to similar families of tree species it is worth asking why there should be a difference. There are several reasons why there could be real differences in the response of forests in the 2 continents. One concerns the length of time over which levels of acidic deposition have been heightened. Another possibility concerns differences between the relative locations of forests on potentially sensitive soils, and of polluting industries.

9.3 Available Models of Tree Response to Air Pollution

Models of tree response to pollutants have been developed at a number of levels, ranging from the assessment of sub-cellular effects on biochemistry and micro-structure to estimation of damage across a continent. The correct level for this study is at the continental level, because of the range of the principal atmospheric pollutants of concern for forest damage.
Forest growth models are made particularly complex by the fact that trees are long lived and need to be managed sustainably. To ensure an adequate supply of timber in future years it is thus important that harvests are properly planned. Even under ideal conditions harvesting levels cannot be suddenly increased beyond a point at which the amount of standing timber starts to fall, without either reducing the amount of timber cut in future years or requiring rapid expansion of the growing stock. If acidic deposition has serious effects on tree growth (which seems likely) it is probable that impacts associated with soil acidification will persist for many years after soils have recovered, whilst the quantity of standing timber recovers to a long term sustainable level.

The following modelling exercises are assessed here:

- Models reviewed by NAPAP;
- The IIASA Forest Study Model;
- The forest module of the RAINS model.

### 9.3.1 Models assessed by NAPAP

The most straightforward models for estimation of the costs of pollutant effects on ecosystems are empirical relationships that link air quality directly to a valuation startpoint, such as reduction in yield. Unfortunately, NAPAP could not identify any such relationships for mature trees. The empirical relationships that do exist tend to relate air quality to a factor such as loss of leaf over a limited period, and even then only for young trees or seedlings. These need to be incorporated into process driven models to allow prediction of valued impacts, or possibly combined with simpler relationships between defoliation and growth. A further problem is that it is generally not possible to check results for anything further than critical loads exceedance leading to increased soil acidity, against observations made in the field, as suitable data does not usually exist.

A number of forest response models were assessed in the NAPAP study. Models varied in complexity and structure. In the summary of the first decade of NAPAP, Kiester (1991) concluded that:

‘None of the models can now be used to produce precise quantitative projections because of uncertainties in our understanding of key growth processes and lack of adequate data sets.’

These models approach the problem in a variety of ways and at a number of different levels of complexity, ranging from the single branch to large areas of forest. Some of the models address damage caused by $O_3$, some acidic deposition, and others both.

A number of specific problems were identified by NAPAP:

1. Projection of effects of pollution on seedlings to mature trees is difficult, given that many dynamic processes of tree growth depend on age and size;
2. Stochastic components of models, such as drought or temperature, may lead to a high level of variation in output;
3. Threshold effects are difficult to model accurately;
4. Interactions, for example between different pollutants, have yet to be properly characterised;
5. Comprehensive data sets, containing all required parameters at a single site, were not available for validation of empirical models;
6. Many models use relationships derived from work conducted on a variety of species in a number of different countries.

These are likely to be applicable to any modelling study of pollution effects on forests. The most serious conclusion, however, was that tree growth in the absence of pollution is not understood particularly well. Fortunately this does not prevent us being able to model pollutant effects on tree growth, though it is likely to mean that there will be gaps in the mechanistic description of pollutant effects.

9.3.2 The IIASA Forest Study model (Nilsson et al, 1991)

Overview

This model is empirical and deterministic in form. It addresses the question of future forest product demand and supply throughout Europe, including Scandinavia but excluding the former Soviet Union. Because of its extensive geographical coverage it has the potential to be very useful in predictive modelling within Europe. The model projects potential biological wood supplies rather than the market supply, as actual harvest figures are often restricted by factors such as roundwood prices, behaviour of forest owners and restrictions for non-wood benefits. Pollution induced decline is simulated using the damage cycles approach of the PEMU model (described briefly below). The study concluded that demand will exceed sustainable supply in Europe by 40 million m\(^3\) by 2010 (in the absence of the effects of pollution), in contrast to a current roundwood surplus of 55 million m\(^3\). When pollution effects were included the deficit increased to 130 million m\(^3\).

Methodology for growth in the absence of decline

Current potential timber supply was assessed on a country by country basis, using data from national forest inventories. To account for differences in forest structure and availability of data within Europe, three approaches were necessary. The first, an area based approach, used descriptions of specific forest types in terms of age and standing volume. Forest type was characterised in terms of species, country, region, owner, forest structure (high forest, coppice, etc.) and site class when suitable data were available. The diameter distribution approach used the same forest type classification but described the state of forests in terms of the distribution of trees by stem diameter. Finally, a simplified approach was used for areas (Greece, Turkey and parts of Yugoslavia) in which insufficient data was available to use either of the other approaches. Growth was estimated from yield table type relationships.

For the purposes of the Forest Study, biological production and amounts removed at final harvest or thinning were estimated under 7 scenarios;
1. Handbook basic scenario. Forests treated strictly in accordance with what is regarded locally as the ideal silvicultural practice. Hence trees are thinned and harvested at times...
that maximise the biological production potential. Comparison of results with actual harvest data suggest the degree to which ideal practice has been implemented.

2. ETTS-IV Basic scenario. Total wood supply was set to the high estimates from the 4th European Timber Trends Study by the UN. This is an indication of the maximum timber supply likely under current practice.

3. Forest study basic scenario. Objective is to strive for consistently high levels of both growing stock and harvest.

4. Forest land expansion scenario. Based on scenario 3, but including scope for expansion of afforested land, to allow comparison with the ETTS-IV scenario.

5, 6 and 7. Scenarios 1, 2 and 3 with pollution-induced decline.

**Decline modelling**

The Forest Study database was combined with the RAINS model ‘to estimate the extent of forest area with depositions in excess of target loads today and in the future’. Deposition estimates for the year 2000 were taken from RAINS based on what were then the current emission reduction plans for SO$_2$ and NO$_x$. Results suggested that in the year 2000 most European forests would still have depositions of S and N in excess of critical loads. Beyond 2000 it was assumed that depositions fell to 0, presumably to highlight the effects of present day emissions.

The PEMU forest decline model (Bellmann and Lasch, 1988; Bellmann et al, 1992) was used to estimate the damages caused by SO$_2$ and N. The model is in 2 parts, the first being a transport and deposition model which will not be discussed here, and the second the pine stand decline model (PSD). PSD first estimates the effects of exposure to SO$_2$ on leaf loss, and then calculates effects on growth and survival.

This approach is an application of results from the Forestry Biomonitoring Programme in East Germany, which investigated the dynamics of pine stands at a number of sites differing in SO$_2$ exposure. 8 sites were established in 1962 around Berlin in an area approximately 175 km square. A characteristic of the study area is the fact that a large quantity of Ca is emitted from industrial sources in addition to SO$_2$ and NO$_x$. Accordingly acid emissions were to some degree buffered by alkaline emissions. Critical levels for SO$_2$ (from Bonsch and Smiatek, 1989) and critical loads for S and N (from Nilsson and Grennfelt, 1988) were exceeded at all sites.

Effects of pollution on growth were calculated by first estimating leaf loss. Two pathways of pollutant impact were identified, a direct (air) pathway and an indirect soil pathway. A compensation factor to account for ‘non-active’ SO$_2$ that had no effect on leaf retention was included in the calculations associated with the soil pathway. The manner in which leaf loss was divided between pathways was not clearly explained in the account read by the present author.

In the development of the model Bellmann et al (1992) state that ‘there seems to be no direct relation between foliage decline and SO$_2$ concentration up to a dose of about 100 µg m$^{-3}$ [about 40 ppb]. It can therefore be concluded that the ecosystems have a compensation
Compensation capabilities are well accepted through the critical levels/loads approach. However, other studies have found significant and negative correlations between SO$_2$ and leaf fall at levels much less than the 40 ppb threshold suggested (e.g. Holland et al., 1991). Indeed, the assumption of this figure for a threshold was based on a single point on a graph of foliage index plotted against pollutant levels. Overall, whilst it is accepted that compensation mechanisms exist, it is felt that the method used to estimate a compensation capability within the PSD model is not satisfactory.

A linear relationship between leaf loss and SO$_2$ was calculated using results from an experiment in which young trees were fumigated within chambers for 3 years at the very high concentration of 800 µg m$^{-3}$ (300 ppb). The amount of time that plants were fumigated during this period is unknown. A mean foliage index score of 4 (on a scale of 1 to 7) was observed at the end of the experiment. The coefficient relating leaf loss to SO$_2$ concentration was then calculated as:

$$\frac{(7-4)}{800} = 0.0037 \text{ [foliage index loss/µg m}^{-3}]$$

This figure was applied throughout to estimate the direct impact of pollutants through the air pathway. This makes a number of assumptions, the most important of which are:

1. The effect was linear from 0 to 800 µg m$^{-3}$;
2. Effects on seedlings can be extrapolated directly to mature trees;
3. Effects on plants growing in enclosed controlled conditions can be extrapolated directly to plants growing in the open under a broad range of conditions;
4. The genetic stock used was representative of that to be found in forests across a wide area of Europe.

Clearly, the choice of relationship is critical in determining the magnitude of the estimated effect. The account given by Bellmann et al. (1992) gives no indication that relationships from other experiments were considered. Analysis of the methods used to estimate the effects of pollutants acting through the soil pathway appears to have been done in a similar manner, using data from the field sites of the Biomonitoring Programme. Nitrogen deposition is regarded by Bellmann et al. as being included implicitly on the basis that the study area was subject to high nitrogen input.

Conclusions on the IIASA Forest Study Model

1. The basic model used in the IIASA Forest Study has a number of features that may make it useful as the basis for a model of pollution damage. The yield table type relationships believed to be used to estimate productivity of forests across Europe model tree growth at an appropriate level of complexity and accuracy. As these relationships are based on historic data it should be possible to check the accuracy of results of the basic growth model. It was also encouraging to note the fact that data were available for every European country, even though the form of these data was not standardised throughout the continent.

2. Serious consideration needs to be taken of the result that showed that timber demand will exceed supply in the next century, even when pollution effects were not included.
3. The method used to estimate damage to non-timber benefits (multiplying the estimate for loss of timber by 2.7, a figure suggested from the work of Metz, 1988, and Stoklasa and Duinker, 1988) is too simplistic. It assumes that all forests will be of equal amenity use, regardless of the species that they contain, their accessibility, and numerous other factors.

4. The decline module is poorly characterised. The relationships come from a single project, with little or no apparent consideration of results from other work. At best it could be applied to the region of Germany from which the damage functions were derived. The implicit assumption of a single mechanism of forest decline and pollutant action throughout Europe is seriously flawed (see Section 9.1). The fact that the data used to derive equations for the model came from an area with high levels of calcium deposition adds a further complication. In a paper describing the model, Bellmann et al (1992) themselves state that the simulations that they present are only valid under the following conditions:
   1. Pine stands;
   2. 40- to 60- year old stands;
   3. Medium nutrient conditions;
   4. Medium site class (average height of stand is about 24 m);
   5. Initial foliar N concentration is in the range 1.2 to 1.8%;
   6. Stand density is equal to one.

9.3.3 The Forest Impact module of the RAINS Model of Acidification (Makela and Schopp, 1990)

Overview

This model is based on the results of a field study in Czechoslovakia, and so, like the PSD model, uses data from a grossly polluted area. In essence, however, the approach is rather more transparent than that of the PSD model, estimating the effects of SO₂ directly from mean air concentrations rather than splitting exposure into air and soil pathways and using a somewhat complicated procedure to calculate the extent to which trees can compensate against pollution. Estimates of loss of production are not given; the model simply estimates the change in risk that results from exposure to SO₂, in terms of the difference between rotation duration and the time taken for a stand to enter a decline phase.

Methodology

Problems associated with the use of the Czech data for modelling work are cited by Makela and Schopp. However, given the way in which the model is applied by the authors these seem to be of little concern for the objectives of their own study. Stress is assumed to be related to altitude and pollutant exposure. Longevity of stands is assumed to be related to total stress.

In the absence of pollution, altitude forms a reasonable guide to the level of stress that a forest experiences; trees at high altitude experience a shorter, cooler growing season than trees lower down, and are also exposed to higher wind speeds and lower temperatures in winter. The
influence of altitude is incorporated into the model as Effective Temperature Sum (ETS), calculated as the sum of mean daily temperatures above a threshold required for growth. This reflects the length and warmth of the growing season. At high altitude ETS accumulates slowly, leading to reduced growth rates and increased rotation times. The increased susceptibility to stress arises partly because of the fact that the trees exist in a more hostile environment, and partly because they are unable to accumulate carbohydrate reserves which under more favourable conditions would normally be available to repair damage.

The dose of SO$_2$, defined as time of exposure multiplied by mean concentration is assumed to describe the integrated effect of the pollutant on tree growth processes in the same way as ETS defines the effect of climate on growth. Effect of dose was assessed through the following framework:

1. Dose accumulates only when a threshold atmospheric concentration is exceeded (introducing the concept of dose compensation that was included in the PSD model);
2. Rate of accumulation is linearly proportional to the atmospheric concentration less the threshold;
3. Damage to trees occurs when cumulative dose exceeds a critical dose;
4. Critical dose is dependent on the Effective Temperature Sum.

Makela and Schopp proposed that stand disintegration will occur once the combined stress of pollutant exposure and ETS was exceeded. This seems acceptable as a simple model of forest dynamics. The effects of pollutants were gauged in terms of the reduction in time taken to reach the critical time at which stand disintegration commenced. The model was applied over Europe using the RAINS grid, in which each cell measures 1° longitude by 1° latitude. Within this grid the distribution of forest was described both in terms of area and altitude. ETS values for each cell were calculated from meteorological data to represent 30 year averages, evaluated at the centre of each grid square for a point at the centre of the square’s altitude range.

**Results and conclusions**

The model correctly identified the areas of Czechoslovakia, eastern Germany and Poland that are experiencing SO$_2$ induced damage. Within this range predictions of stand longevity appear to agree with observations made in recent years. Parts of the Alps, in which SO$_2$ may be influencing tree health, were also identified. The Black Forest was not identified as a risk area. This is consistent with the belief that problems there are not caused directly by high atmospheric SO$_2$ levels, but by acid and nitrogen depositions and other factors instead.

To some extent the model does the same job that comparison of atmospheric pollutant concentrations with critical levels would do: both identify areas at risk. This method appears to be more refined, however, because of the consideration given to the effects of altitude on tree health, a factor known to be significant in several recent decline types. Some estimate of the resulting economic impact could be gained using assumptions about the loss of productivity and non-timber benefits associated with a reduction in rotation time.
To conclude, the model seems suitable for identification of the areas affected directly by high atmospheric concentrations of SO$_2$, though it may not offer a significant advantage over correlation of known deposition with critical levels and loads. It is probably not possible to create a similar model to estimate the areas at risk from O$_3$ damage taking altitudinally related stresses into account as no suitable reference case (necessary for calibration) is known.

9.3.4 Summary of available models

We concluded during the early stages of the ExternE Project that there are no models readily available that permit reasonable assessment of the damages to forests likely to result from fossil fuel pollution. The most ambitious attempt within Europe so far is the IIASA Forest Study Model. Although application of the damage module described is open to criticism, it may at least provide some guide to the order of magnitude of possible effects. It is certainly felt that the basic forest growth module is worthy of further investigation, possibly as the base for a number of species/region dependent damage modules. The forest module of RAINS is well constructed, though appears to be applicable only within a very restricted area.

9.4 Alternative Approaches

9.4.1 Introduction

In the absence of directly applicable models for assessment of the effects of fuel cycle emissions on forests, further work, some of it conducted as part of the ExternE Project, has sought to develop novel approaches to the assessment of forest damage in the last few years. As yet, none of these approaches have been properly validated. However, they demonstrate that statistically significant relationships can be taken from published data. These can be used in the development of relatively simple models that appear to address the problem of pollution induced forest decline at an appropriate level of detail.

The basis of several of these approaches concerns the identification of sensitive areas. This is done by identifying critical loads and levels for different types of ecosystem and mapping these over the area of interest (the whole of Europe in the present case), taking account of variation in soil conditions and the type of ecosystem present. These maps can then be integrated with deposition maps (accounting for both acidifying and neutralizing inputs) to identify the areas where critical loads and levels are exceeded, and hence where the underlying ecosystems are at risk. The following illustrates the application of the critical loads approach to identification of sensitive areas.

Determination of critical loads

Methods to determine critical loads have three levels of sophistication, of which only the first 2 have been applied by the different nations belonging to the UN-ECE (Hettelingh et al, 1991):

- Level 0 which is a ‘semi-quantitative’ approach;
- Level I (steady state modelling); and
- Level II (dynamic modelling).
In the Level 0 approach, soil data presented in map form are used to assign critical loads to ecosystems. The most widely used approach under level I is the steady state mass balance method (SMB). This ignores the time taken to reach a new steady state after a system has been disturbed. It assumes a time-independent equilibrium between the soil solid phase and soil solution. The following assumptions are made using the SMB (UN-ECE, 1990):

- There is no nitrogen fixation or nitrification;
- Net changes in sulphate concentration within the system are negligible (input = output);
- Hydrology and weathering rates can be represented by annual mean characteristics.

To compute the critical load by the SMB method, critical values of chemical parameters are needed. UN-ECE (1990) defined critical chemical values as:

‘the highest value of a critical chemical parameter or combination of parameters that does not cause a significant harmful response in a biological indicator.’

Examples of critical chemical parameters are aluminium concentration, aluminium/calcium ratio, pH and alkalinity (acid neutralising capacity). The critical load of acidity is the difference between the weathering rate of base cations and the leaching rate of alkalinity when the steady state is at the critical point. Therefore, soils on parent rock with a high weathering rate (e.g. limestone) have a higher critical load. The critical steady state is determined by the critical chemical values. The equations to calculate the critical load of acidity are as follows. The critical load of actual acidity, \( \text{Cl}(\text{AC}_{\text{act}}) \), for forest soils is computed by:

\[
\text{Cl}(\text{AC}_{\text{act}}) = \text{BC}_w - \text{Al}_{\text{K}_{\text{crit}}} = \text{BC}_w + \text{H}_{\text{crit}} + \text{Al}_{\text{crit}}
\]

where:
- \( \text{BC}_w \) = weathering of base cations (molc/ha/year);
- \( \text{Al}_{\text{K}_{\text{crit}}} \) = critical alkalinity leaching (molc/ha/year);
- \( \text{H}_{\text{crit}} \) = critical hydrogen leaching (molc/ha/year);
- \( \text{Al}_{\text{crit}} \) = critical aluminium leaching (molc/ha/year).

The critical hydrogen leaching is a product of the runoff \( Q \) and the critical hydrogen concentration, which is assumed to be 0.09 molc/m\(^3\). The critical aluminium leaching is calculated in a similar way if the aluminium concentration is used as critical chemical value. Using the \( \text{Al}/\text{Ca} \)-ratio as critical chemical value the critical aluminium leaching depends on the supply of base cations. In summary, the equations to calculate the critical leaching terms are:

\[
\text{H}_{\text{crit}} = Q \cdot [\text{H}]_{\text{crit}} = 0.9 \cdot Q
\]

\text{Al-criterion:} \quad \text{Al}_{\text{crit}} = Q \cdot [\text{Al}]_{\text{crit}} = 0.2 \cdot Q

\text{Al/Ca-criterion:} \quad \text{Al}_{\text{crit}} = R(\text{Al}/\text{Ca})_{\text{crit}} (\text{BD}^* \cdot d + \text{BC}_w - \text{BC}_a) = 1.5 (\text{BD}^* \cdot d + \text{BC}_w - \text{BC}_a)

where:
- \( Q \) = runoff (m\(^3\)/ha/year);

\[\text{BD}^*\cdot d\] denotes the sum of drainage with base cations.

\[\text{BC}_w\] denotes the weathering of base cations.

\[\text{BC}_a\] denotes the base cations in the parent rock.

\[\text{BD}^*\cdot d\] denotes the sum of drainage with base cations.

\[\text{Al}/\text{Ca}\] denotes the aluminium/calcium ratio.

\[\text{R}\] denotes the ratio of aluminium to calcium.

\[\text{BC}_w\] denotes the base cations in the parent rock.

\[\text{BC}_a\] denotes the base cations in the parent rock.
Impacts of Acid Rain and Ozone on Forests

\[ [H]_{\text{crit}} = \text{critical hydrogen concentration (0.09 molc/m}^3\text{);} \]
\[ [Al]_{\text{crit}} = \text{critical aluminium concentration (0.02 molc/m}^3\text{);} \]
\[ R(Al/Ca)_{\text{crit}} = \text{critical Al/Ca ratio (1.5);} \]
\[ BD^*_d = \text{seasalt-corrected base cation deposition (molc/ha/year);} \]
\[ BC_u = \text{uptake of base cations (molc/ha/year).} \]

In this way two critical loads are calculated, the lower of which is used as critical load of actual acidity.

There are still many difficulties with this approach, as can be illustrated by reference to analysis conducted in Germany. The first German critical loads map of acidity for forest soils was published by Hettelingh et al (1991). Major problems occurred in areas with high annual precipitation because, as the critical alkalinity for forest soils is negative, the critical load increased with higher runoff. For that reason sensitive regions in Germany like the Black Forest or the Harz have high critical loads but these results contradict the present experience (Smiatek, 1992). Further regions where the critical loads in the present map do not reflect the state of the forest are the Bavarian Alps with 39% damaged area but a very high critical load, the Bavarian Forest (40% damaged area, high critical load) and the ‘Thüringer Gebirge’ in the former GDR (50% damaged area, moderate to very high critical load). However, it has to be kept in mind that this comparison implies that acidic deposition is the cause of forest damage in these areas which, of course, is not necessarily true. An improved critical loads map for forest soils in Germany has been prepared where leaching rates are calculated in a modified way.

The exceedance of critical loads

When the load of sulphate, nitrate and ammonium is greater than the critical load, negative effects are expected for forest ecosystems. Base cation and nitrogen uptake by the vegetation effectively increase the load while base cation deposition (which is particularly relevant near the sea) and nitrogen immobilisation decrease it.

The loads below tree canopies are much higher than those measured in the open because tree canopies act as effective filters for trace compounds in the air masses flowing through them (Brahmer, 1990; Klockow and Wintermeyer, 1991). These trace compounds are then ‘washed out’ by rain and cause a sharp increase of total acidity deposited compared to the deposition in the open field. The exact determination of the total acidity deposited below the canopy is complicated by the leaching of compounds from needles, leaves or bark.

To account for the ‘canopy difference’ the wet deposition measured in the open field can be multiplied by filtering factors. The following factors have been used in the present study:

- Sulphate - 1.9 (deciduous), 3.3 (coniferous);
- Ammonia - 1.2 (deciduous), 2.1 (coniferous);
- Nitrate - 1.2 (deciduous), 2.1 (coniferous).
9.4.2 Sverdrup and Warfvinge (1993)

Ulrich’s (1985, 1990) hypothesis states that forest damage is largely due to acidification of the soil giving rise to aluminium concentrations high enough to cause damage to fine roots in the surface layers of the soil. This would in turn affect the ability of forests to take up nutrients and water leading to interactions with stresses such as drought. The hypothesis concentrates on acidic deposition since this is a long term agent of degradation of ecosystems and is widespread across Europe, predisposing trees to damage by other stress factors. Acidic deposition has a cumulative action reducing the buffering ability of the ecosystem. The hypothesis rests on the concept that, as soils acidify, aluminium will be released in the soil solution. Well buffered ecosystems are unlikely to be affected by commonly occurring rates of acidic deposition in Europe. Sites that are already acidic will respond to acidic deposition by increases in aluminium concentrations and will therefore be more sensitive.

It is well known that elevated aluminium concentrations are toxic to plant roots and that it is one of the major determinants of plant growth in acidic environments. The action of aluminium on plant roots depends on the activity of the aluminium ions which, in turn, is related to total ionic strength of soil solution. Thus, the action of aluminium is moderated by concentrations of calcium and magnesium (and other basic ions) and for this reason the Ca:Al ratio, rather than simply the Al concentration alone, is often used as a variable describing aluminium toxicity. Aluminium concentrations are negligible in soil until a low pH level is reached.

Sverdrup and Warfvinge’s (1993) approach follows the work of Ulrich (1985, 1990) to relate the ratio of base cations (Ca+Mg+K) to Al in the soil solution to tree growth. Sverdrup and Warfvinge assimilated a large amount of published data from laboratory bioassays to produce a series of relationship of the form shown in Figure 9.3, which define plant tolerance to the base cation/Al ratio. With regards to the mechanism the paper concentrates almost exclusively on concepts relating to ion uptake. It has been criticised (Kuylenstierna and Chadwick, 1994) for not considering other important conceptual issues, and for ignoring much of the considerable body of early literature which relates species distribution to soil acidity. In particular, concern was noted about the lack of agreement between the assessment of relative tolerance to acidity by Sverdrup and Warfvinge and ecological work on species distribution. Kuylenstierna and Chadwick stated that there is a need for further work to integrate ecological observation into the assessment of relative tolerance.
Figure 9.3 The form of the relationship between base cation/aluminium ration and growth used by Sverdrup and Warfvinge (1993).

An economic assessment of damage using this type of approach was made by Sverdrup et al (1993). Estimates of damage were substantial, though restricted to effects in Norway and Sweden. This approach, as it currently stands, therefore suggests that the externalities of acidifying emissions on forest productivity are likely to be significant.

9.4.3 Kuylenstierna and Chadwick (1994)

The analysis presented by Kuylenstierna and Chadwick is, like Sverdrup and Warfvinge’s work, based on Ulrich’s hypothesis (see Section 9.4.1). The relationship between critical loads exceedance and acidification of the soil provides a potential cause-effect relationship with forest damage.

Data on the health of forests collated by the UN ECE (GEMS, 1988-1991) were compared to deposition and exceedance. The data originate from the International Co-operative Programme on the Assessment and Monitoring of Air Pollution Effects on Forests (ICP forests). The forest damage criterion which has been measured is leaf (needle) loss. In certain countries leaf or needle discoloration have been included. The forest damage statistics are presented as the % of trees in sample plots in a country which are in a certain defoliation class (see Table 9.1).

The UN ECE forest survey results for each country are given in annual reports which provide information concerning the defoliation for conifer, broadleaf and overall tree figures. Statistics for individual species are also reported. Data were collated for the years 1987-1990.
and aggregated as the percentage of trees in classes 1-4, 2-4 and 3-4. Class 1 is not considered in many quarters to represent damage in any form. These data were related to the area weighted average exceedance of critical loads in forested areas in each country, so that they were compatible with the available forestry statistics.

**Table 9.1** Damage classes of the Forest Damage Inventory (BMELF, 1991).

<table>
<thead>
<tr>
<th>Damage class</th>
<th>Description</th>
<th>Leaf loss</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>No damage</td>
<td>&lt;10%</td>
</tr>
<tr>
<td>1</td>
<td>Warning stage</td>
<td>10 to 25%</td>
</tr>
<tr>
<td>2</td>
<td>Moderate damage</td>
<td>25 to 60%</td>
</tr>
<tr>
<td>3</td>
<td>Severe damage</td>
<td>&gt;60%</td>
</tr>
<tr>
<td>4</td>
<td>Dead</td>
<td>100%</td>
</tr>
</tbody>
</table>

A correlation was made between exceedance as a measure of dose and defoliation statistics as a measure of response. The regression analysis indicated that there is a significant relationship between damage classes 1-4 and average exceedance and 2-4 and average exceedance. These show that the amount of defoliation increases with increasing exceedance and indicates that there is a relationship between exceedance and forest damage. However, there is much scatter in the data. This indicates that the regression line is not a particularly good predictor of response to exceedance. This is to be expected, as critical loads exceedance is only one of a number of factors that affect forest performance.

When all species are considered together there would seem to be significant relationships between both deposition, and critical loads exceedance, with degree of defoliation. The relationship between exceedance and degree of defoliation is preferred, on the grounds that the critical loads hypothesis predicts no response below critical load.

The following relationships were derived, all of which relate exceedance to the proportion of trees in each country of Europe for which data were available in damage classes 2-4. All are significant at the 5% level.

- All trees: \( Y = 0.033X + 12.8 \) \( r = 0.52, p = 0.014 \)
- Coniferous only: \( Y = 0.034X + 12.5 \) \( r = 0.49, p = 0.019 \)
- Broad-leaved only: \( Y = 0.027X + 13.2 \) \( r = 0.48, p = 0.03 \)

\( Y = \% \) of trees in damage classes 2 to 4
\( X = \) Average exceedance (meq m\(^{-2}\) yr\(^{-1}\))

These equations may be considered to predict the \% of forest in an area which might be expected to suffer greater than 25\% foliage loss as a result of deposition of sulphur above critical loads. This assertion would only be valid if the cause-effect relationships are confidently established. The low correlation coefficients calculated illustrate the danger of taking this assertion too literally.
For policy analysis the response of timber increment is of greater interest than defoliation. In a recent Swedish study (Söderberg, 1992) defoliation was correlated with results of bore samples taken from a total of 32,325 Norway Spruce and Scots Pine trees. The investigations were carried out for five different regions in Sweden. This work suggested a clear relationship between defoliation and increment for all the regions of Sweden implying that the relationship holds across different climatic regions, from sub-Arctic to temperate. By combination of the relationship between exceedance and defoliation with the relationship between defoliation and increment, a relationship between exceedance and increment may be developed. For illustrative purposes within the present study we have used the relationship between defoliation classes 2-4 for all trees and exceedance of critical loads. This shows a higher degree of significance than other defoliation classes. Also, class 1 is widely considered to show natural variation in crown condition rather than damage, and can hence be ignored.

Within defoliation class 2-4, classes 3 and 4 typically only form 2-5% of the area. This means that most trees lie in defoliation class 2 which represent 25-60% defoliation. The distribution of % defoliation within class 2 is not known. The growth increment decrease for Norway Spruce in this defoliation class was assumed to lie between 10 and 35% and for Scots Pine, 10 and 25%, based on the results of Söderberg (1992). At zero exceedance about 13% of the forest would still be expected to have a defoliation of greater than 25% according to the equations given above. Therefore, only defoliation of more than 25%, in more than 13% of the forest area may be associated with the effect of sulphur deposition in excess of critical loads.

To summarise, the approach goes through the following stages:
1. Quantification of an area weighted average critical loads exceedance for each country;
2. Assessment of the effects of critical loads exceedance on forest defoliation;
3. Assessment of the effects of defoliation induced by critical loads exceedance on timber production.

This approach is clearly only a first attempt to investigate whether data on forest condition collected throughout Europe can be integrated into a simple model of forest response to acidic deposition. An improved analysis would assess relationships between critical loads exceedance and crown condition at a finer level of detail than is currently possible using national average data. Another obvious improvement would be the inclusion of additional factors in these relationships to reduce the high degree of scatter that is currently present - this would become possible once the analysis was brought down to a finer resolution. Use of a finer resolution would also permit checking of results against data collected in the field.

It is necessary to ask what, exactly, the results of this approach tell us. A particular criticism is the fact that explicit account is not taken of the time frame over which a given amount of deposition will affect an ecosystem. Essentially the approach assumes that crown condition in any year is a consequence of critical loads exceedance in that year - a very rough approximation. Overall, because of the failure to account for other sources of variability in the national data on exceedance and defoliation, the approach is likely to underestimate impacts.
In spite of these reservations, this approach was selected for application by the UK team in their report on the coal fuel cycle (European Commission, 1995). It was preferred to other methods because it uses data from field observations collected on a consistent basis throughout Europe, the simplicity of the approach, and the potential for further development work. Further to this, like the work of Sverdrup and Warfvinge, the approach highlights the fact that there is benefit to be gained from examination of the wealth of data that is available for assessment of forest damage.

The methodology used by the German team in the coal fuel cycle report in the current series (European Commission, 1995) was different (see next section). This emphasises the fact that, at the present time, there is no single approach that is obviously better than any other. The good news is that there are a number of alternatives available. What we have sought to do therefore is to explore the use of these alternatives, to provide a basis for subsequent discussion.

9.4.4 Use of the critical loads and levels concept with exposure-response functions specific to German forests

Like the UK team, the German team commenced their analysis by identification of sensitive areas through the application of the concept of critical loads and levels. Given the exploratory nature of the analysis, the availability of data, and the range of applicability of the exposure-response functions used at a later stage of the analysis, the assessment was restricted to Germany (for critical loads) and Baden-Württemberg (for critical levels). Further details are provided elsewhere in this report, and by European Commission (1995).

Correlation of burdens with forest damage inventories

FBWL has tested the correlation between forest damage and pollutant burdens (FBWL, 1989 (see Figure 9.4). The percentage of Norway spruce in damage classes 2-4 in different forest areas in 1986 was used. It was correlated with:

- Total deposition of sulphate and nitrate (throughfall) from 1983 to 1987; and
- Mean 7 h/day ozone concentration during summer (15.4. - 16.10), averaged from 1984 to 1988 (only rural measurement stations with at least three years of measurement have been used).

The correlation between forest damage and total acidity is highly significant ($p < 0.01$), the correlation between forest damage and ozone concentration slightly less significant ($p < 0.05$) while there was no correlation between ozone concentration and acidity. In both cases outlying points were identified. These seemed to confirm that both ozone and acidic deposition need to be accounted for by analysis. In a multiple regression total acidity and ozone concentrations explained 49% of the variation of the forest damage ($p < 0.01$).
Impacts of Acid Rain and Ozone on Forests

Figure 9.4 Correlation between mean total acidic deposition (1983 to 1987) and mean summer ozone concentration (1984 to 1988) and Norway spruce in damage classes 2 to 4 of the FDI 1986 (FBWL, 1989), (Schulze and Freer-Smith, 1991).

The participants of an international forest damage workshop convened under this study in May 1992 agreed that this correlation could be used to estimate forest damage for German Norway spruce forests, though they felt that areas at higher altitudes should be excluded.

Another correlation between forest damage inventory and ozone concentration is provided by Kley et al (1990, see Figure 9.5). Damage classes 2-4 of the Forest Damage Inventory 1986 were again used. These were correlated with the 7 hour per day mean ozone concentration over the period 1984 - 1988 during summer in two steps. First, different forest regions were grouped according to area damaged and the ozone concentration for the forest regions in each group were averaged. Secondly, the correlation between the percentage of damaged forest area and the averaged ozone concentration was calculated. A value for r of 0.81 (p < 0.025) was calculated from the six data points available. Kley et al emphasised that the standard deviation of the mean values is high.

Of course, a statistical relation is not synonymous with proof of causality but an extension of these correlation examples could be a way to develop a rough estimation model for forest damage at least for Germany. In the meantime the data base has improved as two forest damage inventories (1986 and 1991) with a spatial resolution as small as forest region have become available.

Other pollutants and burdens, such as nitrogen enrichment or climatic influences, are omitted by this analysis. The effects are characterised by the damage classes of the Forest Damage Inventory which is a classification by needle loss and yellowing. It is recognised that these symptoms are unspecific and not helpful for the identification of specific decline types and their causes. The influence of the forest damage situation at the starting-point of the analysis as well as predisposing and accumulation effects are neglected.
As for the other approaches identified above, a number of limitations and uncertainties apply to the methodology defined in this section:

- It is restricted to West German Norway spruce forest below 800 m (about 37% of the West German forests);
- There is a high uncertainty associated with the factors for the filtering capacity of the canopy;
- Characterisation of the response (damage classes of the Forest Damage Inventory) is unspecific for the different decline types;
- There are no relationships available for other influencing factors such as nitrogen enrichment, climate, forestry measures, etc;
- Forms of damage other than defoliation are not considered.
9.4.5 Cost analysis based on the application of mitigating measures

Kroth et al (1989) assessed the silvicultural measures which forest managers apply to counteract forest damages, and associated costs for Germany. Table 9.2 shows the total forest areas to which these measures have been applied in West Germany in the period 1988 to 1992. Using the specific costs Kroth et al calculated totals for the whole of West Germany. In the table only those measures which have been approved by experts to have mitigating potential and which are separable from normal operation have been listed. Thus, the total costs for West Germany are in the range of 41.2 to 112.9 MECU/year for the five year period.

Table 9.2 Costs of mitigating measures and forest areas totally affected in West Germany between 1988 and 1992, from Kroth et al.

<table>
<thead>
<tr>
<th>Mitigating measure</th>
<th>Forest area used - 1988 to 1992 (1000 ha)</th>
<th>Costs (ECU/ha)</th>
<th>Total costs for 1988 to 1992 (MECU/year)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Liming and supplementary fertilisation</td>
<td>28 - 230</td>
<td>150 - 500</td>
<td>6.85 - 51.3</td>
</tr>
<tr>
<td>Site mapping</td>
<td>344</td>
<td>33.5 - 35</td>
<td>2.4</td>
</tr>
<tr>
<td>Reforestation of damaged stands</td>
<td>0.3</td>
<td>0 - 5,000</td>
<td>4.95</td>
</tr>
<tr>
<td>Remodelling</td>
<td>1.7 - 4.0</td>
<td>2,500 - 10,000</td>
<td>14.6 - 32.0</td>
</tr>
<tr>
<td>Cultivation of underwood</td>
<td>2.4 - 6.0</td>
<td>600 - 5,100</td>
<td>5.3 - 12.9</td>
</tr>
<tr>
<td>Renovation of protection forests</td>
<td>0.16 mean 30,000</td>
<td></td>
<td>4.8</td>
</tr>
<tr>
<td>Biological protection of forests</td>
<td>not quantified</td>
<td>/</td>
<td>1.75</td>
</tr>
<tr>
<td>Reduction of deer</td>
<td>not quantified</td>
<td>/</td>
<td>0.5 - 2.75 (only state forests)</td>
</tr>
<tr>
<td>Total</td>
<td></td>
<td></td>
<td>41.2 - 112.9</td>
</tr>
</tbody>
</table>

The total figure can be divided by the total area of spruce forest in West Germany subject to critical loads or levels exceedance to provide an estimate of cost per hectare over a five year period. Multiplying this by the incremental increase in area under critical loads and levels exceedance due to operation of the fuel cycle provides a lower estimate of damages, assuming that such measures would be applied. The assessment provides a lower boundary because the analysis is, at the present time, incomplete.

9.4.6 Summary of alternative approaches identified in this study

This section has identified alternatives to the models of forest decline that have been adopted elsewhere, but which we feel fail to provide satisfactory results. None of these alternatives currently provides a completely satisfactory approach to the problem of assessment of forest damage. However, they do explore the use of the enormous volume of experimental and observational data that are available for assessment of forest damage. It is our view that further work in this field should be devoted to the assimilation of existing data, rather than
new experimental work, as we have shown that it is possible to derive new predictive functions from existing data. Further work on modelling should seek to combine some of the different approaches that have been identified here, and any others that may be available.

All of the approaches are linked to assessment of critical loads exceedance. The correlation between critical load exceedance by acidic deposition associated with sulphur and soil pH in Europe shows that this initial condition in the cause-effect relationship holds.

There are indications that defoliation, particularly amongst conifers is related to the rate of acidic deposition and critical load exceedance. Defoliation is of limited interest as a response variable and therefore the relationships between defoliation and increment suggested by the work of Söderberg (1992) are of much significance.

The methodologies presented here are acknowledged to be preliminary. They demonstrate that a link between probable cause (critical loads exceedance for acidity) and effect (reduced timber growth) can be traced using simple data, rather than complex models. It is hoped that this analysis will stir the debate in this area, and identify refinements to the approach (or alternatives) that will allow analysis to be performed with greater confidence.

9.5 Conclusions

1. Fuel cycle pollutants are responsible, or partially responsible for many of the forest declines observed in recent years in Europe. The problem today is different to that in the past, when point sources of pollution were the dominant cause of damage. Trees are now at greater risk from long range transport of pollutants from fossil fuel combustion. Modelling studies have shown that acidic deposition can be carried hundreds of kilometres from the source, affecting areas far removed from any industrial activity.

2. The pollutants of greatest concern to European forestry are SO₂, NOₓ, NH₃, O₃, and acidity.

3. The impact pathway for effects of coal fuel cycle pollutants on forests is complex. A large number of variables, above and below ground will be affected. Knowledge of a number of effects is limited.

4. Direct impacts of acidic deposition on foliage may be limited. It seems likely that the most severe effects result from soil acidification. This can lead to depletion of nutrient base cations from the ecosystem and the accumulation of potentially toxic species, particularly Al.

5. Recent observations of increased tree growth in Europe appear to conflict with concern about forest condition. However, many authorities on this subject believe that the current situation is not sustainable. It appears to result from increased N deposition and increased mobilisation of base cations within the soil. If acid deposition exceeds critical loads (which is the case over much of Europe), ecosystems will become depleted of nutrients. Elevated levels of N will then create nutritional imbalances within plants. This is likely to make them more susceptible to damage.
6. A prerequisite to apply any model is the identification of sensitive forest areas. There are three tools available for this purpose - the forest damage inventory, critical levels exceedance maps and the critical load maps. All have advantages and disadvantages, but good results should be obtained when they are used in combination.

7. There are no models available that predict forest response to pollution, that are both widely applicable and well-accepted by the scientific community. Some models do exist that can be applied over limited areas. NAPAP concluded that the most serious problem in this area was a lack of knowledge of basic forest growth processes. However, it may be possible to circumvent this problem using simpler models based on yield-table and correlative type functions. The problem with these is that they do not explicitly account for the mechanism of pollutant damage. The most ambitious attempt to model the effects of acidic deposition on forests was made by Nilsson et al. (1991), though the decline module used has been heavily criticised. The basic design of the model, however, may well be useful as a framework into which decline modules can be inserted in the future.

8. The main problem for modellers is the fact that little has been done to develop dose-response functions for trees. Also, through economic necessity, most experimental work has been conducted on seedlings or saplings. Few studies have assessed the response of mature trees. This is important because there are numerous reasons for tree response to vary with age. We have shown that a variety of exposure-response functions are available for analysis of pollutant effects on forest condition and growth. Some of these are derived from experimental data, others from analysis of data collected in the field. Further assessment of available data should be treated as a priority.

9. Valuation of lost timber is relatively straightforward, though results are, of course, highly sensitive to discount rate, given the longevity of both trees and ecosystem acidification. Little has been done concerning valuation of non-timber values of forests. Nilsson et al. (1991) suggested the use of a multiplier (of 2.7) of damages through lost timber production. This is clearly a preliminary judgement in the absence of better data.

10. Measures which forest managers can use to mitigate new types of forest damage have been described. A possible approach to integrate the costs for those mitigating measures into the assessment has been outlined. This assessment depends on the existence of a forest damage model and knowledge of the probability of a mitigating measure being applied. However, at this stage it seems doubtful as to whether sufficient information is available to connect mitigating measures to power plant emissions.

11. Overall it is concluded that there is good reason to believe that fossil fuel cycle damages on forests could be significant. A variety of approaches have been demonstrated in the present study. Whilst none of these are held up to be a definitive methodology for the problem, they do show that analysis is possible. It is hoped that the use of these methodologies will stimulate debate, and that this will lead to further advances in the modelling of pollution effects on forest decline.
9.6 References


Impacts of Acid Rain and Ozone on Forests

Fumigation Experiment. Contract Report to National Power Technology and Environment Centre, Leatherhead UK.


Impacts of Acid Rain and Ozone on Forests


Smiatek, G. Personal communication of G. Smiatek, Institute of Navigation, University of Stuttgart, April 1992


10. EFFECTS OF ACIDIFICATION ON RECREATIONAL FISHERIES

10.1 Introduction

Much of the early concern associated with the growing public awareness of ‘acid rain’ as an environmental problem focused on the effects on fish stocks in rivers and lakes. In Europe the problem has been most serious in the northern countries, particularly Scandinavia and Scotland. This is partly the result of the prevailing wind direction with respect to sources of acidifying pollutants and partly a consequence of the underlying geology in these regions, which is largely composed of hard rocks that weather slowly. The resulting slow release of base cations is insufficient to neutralise current acidic inputs over a large area.

Virtually all of the water that enters rivers and lakes falls first as rain onto land, before draining through the soil into rivers and lakes. Drainage water chemistry is controlled by the interaction of a number of factors of which the most important are precipitation chemistry and quantity, vegetation, soil, geology and hydrology. The impact of acidic deposition on drainage water chemistry is controlled by complex interactions. The dominant controls at a catchment or regional scale are the characteristics of the soils and bedrock, and in particular soil and bedrock chemistry and mineralogy.

A considerable amount of evidence is available demonstrating changes in the chemistry of surface waters in response to inputs of acidic deposition. This evidence includes regional surveys of water quality, detailed process studies at specific sites (Christophersen et al, 1982), experimental additions of acid to catchments, removal of acidity from atmospheric inputs (Wright et al, 1988) and reconstructed historical records of water chemistry for lakes using diatom records (Renberg and Battarbee, 1990).

Significant impacts of acidic deposition on drainage water chemistry are only usually found in areas with acidic soil and bedrocks with a small buffering capacity. It is in such areas that reductions in fish populations have been reported. The most important group of fish in the surface waters of these areas in western Europe are the salmonids, the group of fish that includes salmon and trout.

Clear relationships between water chemistry and the survival and breeding of fish have been demonstrated using laboratory experiments, manipulation of stream water chemistry, and linked regional surveys of fish populations and water chemistry. For example, salmonid survival and breeding has been shown to be highly correlated with concentrations of hydrogen, aluminium and calcium in low conductivity waters (Howells et al, 1983). Research over the last 25 years has also demonstrated major reductions in fish populations in certain areas which have been impacted by acidic deposition e.g., southern Norway and Sweden, Scotland, Wales and south-east Canada (Munitz and Walloe, 1990).
Of course, from an ecological perspective, fish are only one element of an aquatic ecosystem. However, their health can be taken as an index of the quality of the system as a whole. The fish populations of surface waters are determined by the interaction of a number of factors which include water chemistry, food availability, water temperature and predation. To assess impacts of acidification on fish, it is therefore necessary to use a series of complex models. The methodology presented here has been developed through discussion with an international panel of experts convened under the ExternE Project.

An impact pathway for effects of acidic deposition on fisheries is shown in Figure 10.1. As in other cases, all known effects, including feedbacks have been included, whether or not these are thought to be quantifiable at the current time. The comprehensive nature of these pathways is intended to allow the effects that have been quantified to be put into perspective with those that have not. Consideration of all potential impacts will also assist in the identification of priorities for future research.

For the issue of acidic deposition and fisheries, an additional factor can be considered - the expenditure on measures to mitigate against damage. Approximately 30 Million ECU is spent each year on liming lakes, rivers and catchments in Norway and Sweden, to temporarily reduce the effects of acidic deposition (Hanneberg, 1993). The acidity which produces this damage arises from a number of individual countries. It is thus possible to apportion these total costs to the quantity of acidity deposited in Norway and Sweden which originates from individual countries emissions. However, it must be stressed that liming has a temporary effect, and is not a real solution to the problem. Moreover, it is not applied in all areas in which fish populations have declined.

In order to follow the damage function methodology, a more rigorous approach is required. This quantifies the impacts on fisheries by evaluating each stage from emission through to effect. Because of the lack of direct relationships between changes in atmospheric deposition and fish populations, it is necessary to link cause and effect using hydrochemical models, which relate changes in precipitation chemistry to changes in surface water chemistry, and other models relating fish populations to water chemistry. These models are discussed in the following section.
Effects of Acidification on Freshwater Fisheries

Figure 10.1 Impact pathway for acidic deposition effects on freshwater fisheries.
10.2 Linking Hydrochemical and Fish Status Models

As fish populations are determined by a number of factors, it is necessary to assess impacts of acidification using a series of models and databases. The following models and databases are required:

Models
1. A deposition model which calculates atmospheric deposition of major solutes. This data can be used as an input into the selected hydrochemical model;
2. A hydrochemical model which calculates stream water chemistry, including major anions and cations. This data can be used as an input to the selected fish population/fishery status model;
3. A fish population/status model which links population/fishery status and water chemistry.

Databases
1. Modelled deposition data for major cations and anions;
2. Catchment characteristics - soils, vegetation, hydrology;
3. Stock at risk - number of lakes/streams or length of streams and lake/streamwater chemistry.

10.2.1 Hydrochemical models

The simplest type of hydrochemical model is an empirically derived relationship based on current data on stream/lake chemistry, which can be used to extrapolate from present conditions. For example, Henriksen (Henriksen et al., 1988) derived a steady state model which assumes that water chemistry is in equilibrium with atmospheric inputs and which is based on balancing the strong acid anions and sources of alkalinity in solution. The acid neutralising capacity of the water body at differing inputs of acidity, under steady state conditions, can be calculated. The limitations of the model are the steady state assumption and the limited range of output variables which can be used as input to a fish density/fishery status model.

A number of process based dynamic hydrochemical models have been developed which can be used to determine drainage water chemistry at a given site using precipitation chemistry as a driving variable, e.g. MAGIC (Cosby et al., 1985), SAFE (Warfvinge and Sverdrup, 1990) and RAINS (Alcamo et al., 1990). These models incorporate the main processes controlling water-soil and water-rock interactions (e.g. mineral weathering, ion exchange, sulphate adsorption) and combine them with a hydrological model of varying complexity. Some of the models also incorporate plant uptake and decomposition processes. The input data required to run the models varies but generally includes information on atmospheric inputs, precipitation amount, runoff, soil chemistry, coefficients for weathering rates, sulphate adsorption and aluminium dissolution, and data on net uptake of plant nutrients. Output from the models includes concentrations of the major solutes in the stream waters. Most of the models work in annual time steps.

The various models have now been applied to a wide range of sites in western Europe and in North America. Perhaps the most widely used of the models has been MAGIC which has now been applied to catchments in at least ten different countries. Validation of the model outputs is
very difficult, especially when the models are used to predict trends in drainage water chemistry over tens of years into the future. The most effective test is provided by manipulative experiments, which have indeed been performed for MAGIC, showing that the model reproduced the main trends in observed data. These and other results suggest that the models can be used with a reasonable degree of confidence.

10.2.2 Stock at risk

The stock at risk can be considered as the total population of streams, lakes or fisheries within the country or region of interest, or that part of the population occurring within areas considered to be sensitive to the impacts of pollutant deposition. If the latter approach is used, a method is needed of defining and delimiting the sensitive areas. A number of such schemes have been developed which use information on soils and/or geology (e.g. Hornung et al, 1990 and 1995; Holmberg et al, 1987; Hamm et al, 1987; Norton, 1986). An example of the output from such a scheme, plotting moderate to high sensitivity areas of surface water in the UK, is shown in Figure 10.2.

Whether the total population, or the population in sensitive areas only, is considered, data on stream/lake chemistry and fishery status is required for a sample of streams or lakes; it will rarely, if ever, be possible to obtain the required information for the total population. The size of the sample will clearly influence the reliability of the population estimates. Fishery data can be expressed as fish density, number of fish per unit of stream or lake surface area (e.g. Ormerod et al, 1990), or in terms of the status of the fish population, for example Henriksen et al (1989) and Rosseland et al (1980) used three classes - healthy, marginal and extinct - to describe fish populations in a survey of lakes in Norway. Surveys of fish populations are usually based on electro-fishing or catch removal methods; chemistry is determined for samples collected from the same streams.

10.2.3 Models of the effects of acidification on fish populations

A number of statistical relationships, or models have been derived for calculating acidification effects. Ormerod et al (1990) used data from a 1984 synoptic survey of fish populations and water quality of 83 headwater streams in Wales. The data has been used to develop empirical models relating trout survival and trout density (per unit area of stream) to water quality and flow. Separate relationships can be developed for pH, aluminium and hardness. The model which explained the largest proportion of the variation incorporated both aluminium and hardness:

\[
\log_{10} D = -1.24 - 1.08\log_{10} Al_{\text{filt}} + 1.33\log_{10} \text{Hardness} - 0.22\log_{10} ADF
\]

where;
- \( D \) = density as number of fish / 100 m\(^2\),
- \( Al_{\text{filt}} \) = Aluminium in mg l\(^{-1}\),
- \( \text{Hardness} \) = mg CaCO\(_3\) l\(^{-1}\),
- \( ADF \) = average daily flow in m\(^3\)s\(^{-1}\).
Figure 10.2 Predicted sensitivity of surface waters in England, Scotland and Wales to acidic deposition inputs.
Harriman et al (1990) derived a relationship between stream pH and fish numbers (brown trout) per unit area of stream based on a study of some 20 streams in Scotland. The resulting regression is very similar to that derived for pH and fish density from the Welsh data. This suggests that the models are relatively robust and could probably be applied in Wales, Scotland and northern England.

Munitz and Walloe (1990), Bulger et al (1993) and others have developed regression models linking lake chemistry and fishery status in Norway, based on the results from surveys of a large number of lakes. The model derived by Bulger et al showed that a regression incorporating pH and labile aluminium explained between 52 and 64% of the variance in trout status. Bulger also explored more complex statistical approaches based on discriminant function analysis; the application of the simple regression and the discriminant approach have been explored in a Norwegian context by Cosby et al (1994).

10.2.4 Linked models

Cosby et al (1994) have recently linked the Henriksen empirical water chemistry model and MAGIC hydrochemical model to the statistical models relating status, and change of trout population to water chemistry, that were developed by Bulger et al (1993) for Norwegian lakes. The Henriksen model was linked to fish response models based on ANC while the MAGIC model was linked to fish response models based on:

(i) All major ions; and
(ii) All major ions plus ANC.

The fish population status and change of status classes, from the original 1,000 Norwegian lakes survey (Henriksen et al, 1988 and 1989) were combined to give four resultant classes; healthy, marginal unchanged, marginal declined and extinct. The linked models have been used by Cosby et al (1994) to predict the variation in the proportion of the population of lakes in the different status/change of status classes at a range of sulphur deposition scenarios.

Ormerod et al (1990) have used a similar approach in the UK, linking the MAGIC hydrochemical model to empirical models of survival and density of brown trout populations (Figure 10.3). The empirical model on trout density relates trout density, per unit area of stream surface, to water quality and flow and was based on data from a regional survey of water quality and fish populations in Wales.

10.2.5 Valuation issues

Valuation of changes in fish populations, the end point quantified by the methods just described, has not been performed so far within ExternE. The results of a contingent valuation study that has recently been completed (ECOTEC, 1994) may be useful. However, there appears to be no other data that could be used from previous studies.
There are a number of aspects that should be covered by the valuation (see Figure 10.1):
- Recreative value;
- Commercial value;
- Existence value;
- Costs of any mitigating measures taken.

The last of these is covered in Section 10.3. Assessment of recreational and commercial values requires knowledge of the behaviour and value systems of anglers, naturalists and others who make use of freshwaters because of their biological value.

In the absence of a means of applying a full valuation of impacts, the analysis conducted so far within the ExternE project has been limited to a fraction of the area affected by emissions from a power station (see Section 10.2.8). Extending the analysis to the total affected area is possible, though there are a number of important uncertainties described below.

**Figure 10.3** A schematic representation of the modelling procedure applied to Welsh catchments (after Ormerod *et al*., 1990).
10.2.6 Uncertainties and limits on the application of the approach

One of the most important uncertainties in such studies results from the current decline of emissions and deposition of acidifying pollutants in Europe - there is no firm data on future emission levels. It is necessary, therefore, to rely on estimates of future emissions and modelled deposition patterns.

A number of other issues relate to the MAGIC model. The model does not incorporate all processes which influence streamwater chemistry. The weathering rate is derived using an optimisation routine. Soil input parameters are either based on the most widespread soil type in a catchment or on weighted means. However, the model has been validated against data from a number of sites in different countries (see above), and has been found to be reasonably robust.

The fish density model is a statistical model which explains some 50% of the variation in trout densities in the sample of streams from which it was derived. It would of course be expected to explain less of the variation among fish populations outside of the sampling area, though it appears reasonable to use it in other parts of the UK (see above). Data is also available for the Scandinavian countries, though some effort may be required to adapt this to a form amenable to the style of assessment described here. Far less data is available for sensitive areas in other parts of Europe. However, the main stocks of fish at risk are to be found in the northern countries for which data is available. Extrapolation of the analysis to other sensitive areas may thus not introduce too serious an uncertainty in terms of the overall analysis of effects on fish populations.

10.2.7 Demonstration of the methodology

The application of the approach described here is demonstrated in the ExternE coal report, for the West Burton ‘B’ case study (European Commission, 1995a). Given that valuation was not possible the analysis was performed over only a small area of Wales, to illustrate the methodology.

The full impact of any increased deposition of acidic pollutants on water quality, and hence on fish populations may not occur for a number of years after the increase begins. The analysis undertaken so far shows that if the MAGIC model is run forward for a period of 40 years with the effect of the incremental deposition included, aluminium levels in streamwater increase gradually over time. Ideally the predicted stream water chemistry for each year should be used as input to the trout density model.

10.3 Mitigation Costs

A substantial amount of money (30 MECU) is spent annually in Norway and Sweden to apply lime to counteract atmospheric inputs of acidity. Most of this money is spent in an attempt to maintain or improve the quality of freshwaters, though some is also used in forestry. For the purpose of our analysis there is no need to separate treatment of rivers and lakes, and forestry, though the costs associated with liming forests should not be double counted in other parts of
the analysis. It must be noted that liming is not a solution to the problem; it is viewed in Scandinavia as a means of ‘buying time’ whilst acidic deposition is reduced to an appropriate level following international agreement on emission reductions.

The EMEP transfer matrices (EMEP, 1992) have been used to estimate the incremental amount of acidity that would be deposited in Norway and Sweden from several of the fossil fuel fired power plants considered so far by ExternE (European Commission, 1995a, b). This was expressed as a fraction of the total deposition in Norway and Sweden (separately). The result was then multiplied by expenditure on liming in each country.

The use of the transfer matrices does not allow differentiation in effect between high level (e.g. power stations with tall stacks) and low level (e.g. traffic) sources of acidity. For this reason it seems likely that the effects of power stations in the UK and Germany are probably underestimated, as high level emissions are likely to travel further than low level emissions. It is anticipated that this problem will be resolved in the next stage of the ExternE Project following recent improvements made to the Harwell Trajectory Model.

Mitigation costs should only be included if there is a reasonable likelihood of mitigating action being taken. If there is little or no likelihood of increased expenditure on mitigation the analysis should be restricted to assessment of the damage associated with the incremental emissions from the plant in question. In the present case it is possible that expenditure on liming in Norway and Sweden would not increase beyond the current figure of 30 MECU - in other words that this represents the most that the governments of the two countries are either willing or able to pay to alleviate acidification through liming. Is it therefore logical to ascribe additional costs for liming for an incremental investment in power projects that would lead to increased acidic deposition in these countries? Would the effect of a marginal increase in deposition be reflected instead solely through a marginal increase in damage to fish stocks and forests? We believe that it is valid to include expenditure on liming, on the grounds that international agreements will lead to a reduction in deposition in the coming years. Investment in technologies that emit acidifying pollutants may therefore reduce the rate at which expenditure on liming could be reduced to reflect reductions in deposition.

10.4 Summary

A sophisticated analysis of the effects of acidic deposition on the fish populations of rivers and lakes is possible using the methods outlined in this Chapter. The analysis links together a series of models describing effects of deposition on water chemistry and resulting impacts on freshwater biology. Within the ExternE Project so far the application of these models has been limited to assessment of the impacts of the West Burton ‘B’ coal fired power plant (European Commission, 1995a) within a small area of Wales. However, it seems likely to be directly applicable in many other acid-sensitive areas of the UK. Use in Scandinavia requires alteration to the fish dose-response model, though not to the MAGIC model which has been validated in many countries. Given the vast quantity of data that has been collected in Scandinavia it seems at least possible that a suitable model could be derived with relatively little effort, based on the work by Ormerod et al (1990). Application in other acid-sensitive areas of Europe seems likely to be more difficult.
Although valuation of anything other than mitigation costs for the liming carried out in Norway and Sweden has not been possible, we have shown that it is possible to quantify impacts not just in terms of exceedance of critical load, but also in terms of changes in fish populations. This provides a starting point for future valuation studies. In any case, it has been demonstrated that it is possible to go beyond critical loads exceedance, or assessments of numbers of acidified lakes and streams, to provide impacts in terms which may be more amenable to policy analysis, even if valuation cannot be carried out for the foreseeable future.

The following areas for future research were identified by the international expert panel convened under the study. They fall into 3 main groups, relating to the modelling of water quality, assessment of biological effects and valuation issues.

**Modelling water quality**
- Improved models of base cation deposition.

**Assessment of biological effects**
- Assessment of the relationship between critical load exceedance and the biological impact.
- Dose-response models for areas outside the UK and Scandinavia.
- The time function for fish response to changes in water quality.
- Dose-response models for migratory fish with relationship to water quality.
- Scaling up from models derived for nursery streams to larger stream and river systems.

**Valuation issues**
- Relationships between water quality and the size of the exploitable resource.
- A definition of the sustainable resource for recreational fisheries and a definition of yield.
- Relationships between fishing activity and fish numbers/populations.
- Time functions for responses of ‘willingness to fish’ in response to changes in fish numbers.

Clearly the most outstanding requirement is for an agreed valuation procedure to link in with the endpoint of the analysis; the estimated reduction in fish numbers.
10.5 References


Effects of Acidification on Freshwater Fisheries


11. IMPACTS OF AIR POLLUTION ON BUILDING MATERIALS

11.1 Introduction

The effects of atmospheric pollutants on buildings provide some of the clearest examples of damage related to the combustion of fossil fuels. Pollution related damage to buildings includes discolouration, failure of protective coatings, loss of detail in carvings and structural failure. The rate at which these types of damage occur has increased greatly since the Industrial Revolution (Feenstra, 1984). A good example is provided by some natural stone walls at Cologne Cathedral, some of which were enclosed when the building was finished in the 19th century, following exposure to ambient conditions for several centuries. The parts of the walls that are now inside the Cathedral are still in excellent condition, whilst those that remained outside have deteriorated seriously. The damage is therefore attributable to pollution over the last 100 years (Efes and Luckat, 1976).

Impacts of acidic deposition on materials are, of course, not restricted to buildings of cultural value. They have also been recorded on modern buildings and to other types of materials such as textiles, leather and paper. Given the relative abundance of modern buildings compared to older ones, it may be anticipated that damages to the former will outweigh those to the latter.

However, valuation of material damage is complex for a number of reasons. It is highly dependent on the material and the cultural significance of the object in question. Replacement and maintenance costs are probably the easiest to evaluate provided that there are clear guidelines as to what action should be taken and at what time (e.g. replacement of steel when a given depth of material has corroded).

For buildings of aesthetic or cultural merit, such as ancient cathedrals, estimating the effects of damage is extremely difficult. Consideration should be given to the amenity and existence values for such buildings, as replacement costs do not adequately reflect the full cost of, for example, the loss of fine carvings that are several hundred years old. Costs associated with damages of this type are extremely site-specific, not only in terms of the merit of the item in question, but also in the way in which it can be treated. A further problem is that there is currently no available inventory to describe the stock at risk.

This chapter presents a discussion of the methodology used in the assessment of material damages within the ExternE Project. We have not attempted to produce a complete assessment of all impacts on materials. The extent to which each category of damage has been analysed is partly a reflection of the expected magnitude of associated externalities, and partly a reflection of the current state of scientific knowledge. The approach draws on a combination of the methods used by individual project teams and uses the most up to date
Impacts on Materials

data available. However, no one approach or set of dose-response functions can be regarded as definitive and instead we have taken a pragmatic approach, trying to present a consistent and unified method for analysing the effects of air pollutants on materials.

11.2 Impact Pathways to Describe the Effects of Fossil Fuel Cycle Pollutants on Materials

We have identified four impact pathways which describe the effects of acidic emissions, and precursors of photo-oxidants, on stone, metals, polymeric materials (paints, plastics and rubbers) and fine art materials. These pathways are shown in Figures 11.1 to 11.4. The overall design of the impact pathways is identical to that used for impacts on ecosystems.

Accordingly they are divided into the following sections:
I. Emissions of acidic pollutants and precursors of photo-oxidants;
II. Atmospheric transport and chemistry;
III. Deposition processes;
IV. Intermediate processes;
V. Physical/chemical startpoints;
VI. Physical/chemical endpoints;
VII. Valuation.

A detailed impact pathway for the effects of particulate emissions on building soiling is not necessary, though some analysis of this effect is included towards the end of this chapter.

11.2.2 Extent of pathway implementation

The impact pathways below show that for most materials, impacts fall into three categories:
• Discoloration;
• Material loss;
• Structural failure.

There are no valuation studies or material inventories from which estimates of the costs of discoloration can be estimated. However, such effects seem likely to be small. Structural failure resulting from pollutant exposure seems unlikely unless either the design of a building is fundamentally flawed, or the owner of a property has not carried out routine maintenance.

Therefore, the analysis presented here of material damages is limited to the effects of acidic deposition on corrosion. Acidic deposition covers both the direct effects of sulphur dioxide and the effects of acidity resulting from both SO₂ and NOₓ emissions. In addition, for some fuel cycles, a simple method has been used to quantify the deposition of particulates on building soiling.
Figure 11.1 Impact Pathway for the Effects of Acidic Deposition on Stone.
Figure 11.2 Impact Pathway for the Effects of Acidic Deposition on Metals.
Figure 11.3 Impact Pathway for the Effects of Acidic Deposition on Polymeric Materials (Paints, Plastics and Rubbers).
Figure 11.4 Impact Pathway for the Effects of Acidic Deposition and Photo-Oxidants on Fine Art Materials.
It should be noted that the effects of these air pollutants are set against a background of substantial natural weathering forces including rain, bacteria, freeze-thaw cycles and sea salt (in coastal regions). These natural constituents would lead to damage of materials even in the absence of air pollutants. However, measured deterioration rates are a factor of 10 to 100 lower than in the presence of air pollutants.

For a number of materials, the dry deposition of SO₂ exerts the strongest corrosive effect of atmospheric pollutants. Wet deposition of pollutants, expressed as rain acidity, has a corrosive effect on certain materials but is generally weaker. The role of atmospheric NO₂ has not yet been clarified. Although a strong synergistic effect with SO₂ has been observed in laboratory studies, this has not yet been observed in the field.

Ozone is known to damage some polymeric materials such as paints, plastics and rubbers (Lee et al., 1994). It has also recently been observed to act synergistically with SO₂ in the field (Kucera et al., 1993a; Kucera, 1994). However, within this phase of the ExternE Project, the damage caused by incremental ozone levels resulting from fossil fuel emissions has not been considered. Further research is required, both on the models used to predict tall stack emissions on O₃ levels, and on the effects of O₃ on materials, before these impacts can be quantified.

The damage assessment so far within the ExternE Project has been undertaken for ‘utilitarian’ buildings (houses, shops, factories, offices and schools) using a repair cost method. For the analysis of UK power plants, we extended this analysis to galvanised utilitarian structures in other sectors, including agriculture, transport (e.g. street furniture and railway gantries) and energy (e.g. electricity pylons). Utilitarian structures of other materials are not considered because no suitable inventory exists. Buildings of aesthetic or cultural merit, such as ancient cathedrals, have not been assessed for the consequential damage from impacts, as there is insufficient data concerning material inventory, repair and existence values for such buildings.

The materials for which damage have been considered are calcareous stone, mortar, paint, concrete, aluminium and galvanised steel. Although not exhaustive, this list includes the most sensitive of the materials commonly used by the construction industry. All steel is assumed to be painted and this stock is therefore transferred into the paint inventory. A summary of the materials and their susceptibility to air pollution is shown in Table 11.1.

No consideration has been made of the loss of transparency for glass from the effects of fossil fuel cycle emissions. Modern glass is considered to be very resistant to attack and therefore the impacts of power plant on glass in utilitarian structures should be negligible. This is not true of older glass, particularly when it is coloured, painted or stained. However, as in other cases assessment of damages to non-utilitarian structures and materials is not currently possible because of the lack of inventory and valuation data. There is concern that some novel grades of glass produced for specialist markets may be more susceptible to pollution attack than other modern glass (Fuchs, 1994). Such glass is becoming increasingly popular. There is again, however, insufficient data available for investigation of this issue.
Table 11.1  Sensitivity of Materials to Air Pollution and the Stock-at-Risk in Europe.

<table>
<thead>
<tr>
<th>Material</th>
<th>Sensitivity to air pollution</th>
<th>Stock-at-risk in Europe</th>
</tr>
</thead>
<tbody>
<tr>
<td>Brickwork</td>
<td>uncertain</td>
<td>very large</td>
</tr>
<tr>
<td>Concrete</td>
<td>low</td>
<td>very large</td>
</tr>
<tr>
<td>Natural stone (sandstone, limestone, marble)</td>
<td>high (severely affected by SO₂)</td>
<td>large (especially culturally valuable objects)</td>
</tr>
<tr>
<td>Unalloyed steel</td>
<td>high (severely affected by SO₂)</td>
<td>very small</td>
</tr>
<tr>
<td>Stainless steel</td>
<td>very low (excellent resistance to pollutant attack)</td>
<td>medium</td>
</tr>
<tr>
<td>Nickel and nickel-plated steel</td>
<td>high (especially in SO₂-polluted environment)</td>
<td>very low</td>
</tr>
<tr>
<td>Zinc and galvanised steel</td>
<td>high (especially in SO₂-polluted environment)</td>
<td>medium</td>
</tr>
<tr>
<td>Aluminium</td>
<td>very low</td>
<td>medium</td>
</tr>
<tr>
<td>Copper</td>
<td>low (formation of patina layer in polluted environments)</td>
<td>low</td>
</tr>
<tr>
<td>Lead</td>
<td>very low (one of the most resistant materials)</td>
<td>low</td>
</tr>
<tr>
<td>Organic Coatings</td>
<td>uncertain</td>
<td>very large</td>
</tr>
</tbody>
</table>

Finally, there are impacts from particulate emissions on buildings. The most obvious of these is the discolouration of stone and brickwork. Recent major reductions in urban smoke emissions have significantly reduced such soiling impacts, but the problem remains. We present a simple top-down approach for the quantification of these damages.

11.3 Assessment of Damage to Building Materials

To quantify the corrosion impacts on building materials from acidic deposition, it is first necessary to establish the ‘reference’ environment. This includes defining the quantity of material present, known as the ‘stock at risk’, and the meteorological and ambient pollution levels. As acid deposition is a regional scale phenomenon, the reference environment must be defined on a European level.

11.3.1 Quantifying the stock at risk

The stock at risk is derived from data on building numbers and construction materials taken from building survey information. Such studies are generally performed for individual cities; these can then be extrapolated to provide inventories at the national level. In cases, where individual country data is not available, values must be extrapolated from other countries although this may result in a loss of accuracy. A discussion of how the stock at risk was built up for the assessment of UK and German analysis is presented below.
The compilation of UK building identikits

For the calculation of the UK ‘stock at risk’, the bulk of the data was derived from a detailed study of building materials damage in the city of Birmingham (ECOTEC, 1986). This study identified 21 common building types, and using field research calculated the average size and amount of construction materials making up the external envelope for each building type in the city. Due to limited statistical data being available for the UK as a whole, the 21 common building types identified by ECOTEC were reduced down to 5, in order to obtain estimates for the much wider building types. For example, ECOTEC provided separate ‘identikits’ for convenience goods shops, durable goods shops, light commercial shops and department stores. In the aggregation, the proportion of these types of shops were assumed to be that given for Birmingham (ECOTEC, 1986). From this it was possible to calculate an aggregate identikit, which could then be used to represent all shops in each grid square.

The five building materials 'identikits' were:
1. Dwellings;
2. Schools;
3. Shops;
4. Offices;
5. Industrial buildings.

An example of one of these model building descriptions, is shown in Table 11.2. The other four identikits for schools, shops, industrial buildings and offices are shown in Appendix 11.1.

In the absence of detailed information for the UK as a whole, the identikits for Birmingham were taken to be representative for the UK. This assumption is not unreasonable. Birmingham is the second largest city in the UK with a population of over one million people. It has a wide range of industry, such as engineering, cars, light industry and services and a varied housing sector, from high rise flats to large detached housing. The main sources of error are likely to be the under-estimation of rural building types and therefore the quantity of natural stone present. To compensate for this, an upper estimate of the quantity of stone has been taken from other European building surveys. This approximately doubles the stock of stone at risk.

The compilation of UK building distribution

The next stage of the reference environment compilation was to estimate the number of each building type across the UK. For the purposes of this study, we used a spatial resolution defined by the 100 km x 100 km grid cells of the UK National Grid.
Table 11.2 Building Identikit 1 Used to Describe the Use of Different Materials Used in the Construction of Houses for the UK Implementation.

<table>
<thead>
<tr>
<th>Building identikit 1</th>
<th>Dwellings</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flat Roof Area (m²):</td>
<td>12.79</td>
</tr>
<tr>
<td>Pitched Roof Area (m²):</td>
<td>54.3</td>
</tr>
<tr>
<td>External Wall Area (m²):</td>
<td>112.88</td>
</tr>
<tr>
<td>Windows and Doors (m²):</td>
<td>11.87</td>
</tr>
<tr>
<td>Drainage (m²):</td>
<td>10.62</td>
</tr>
<tr>
<td>Total External Envelope (m²):</td>
<td>202.46</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Predominant materials</th>
<th>Model (%)</th>
<th>Quantity (m²/building)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flat Roofs:</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Felt on Garages</td>
<td>0.52</td>
<td>6.63</td>
</tr>
<tr>
<td>Asbestos Cement Tiles</td>
<td>0.17</td>
<td>2.21</td>
</tr>
<tr>
<td>Asphalt on Screed</td>
<td>0.31</td>
<td>3.95</td>
</tr>
<tr>
<td>Pitched Roofs:</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Metal/Galvanised Steel</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Clay Tiles</td>
<td>0.24</td>
<td>13.31</td>
</tr>
<tr>
<td>Slate Tiles</td>
<td>0.41</td>
<td>22.14</td>
</tr>
<tr>
<td>Concrete Tiles</td>
<td>0.35</td>
<td>18.85</td>
</tr>
<tr>
<td>Wall Area:</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Concrete Panels</td>
<td>0.01</td>
<td>1.14</td>
</tr>
<tr>
<td>Concrete Frame Elements</td>
<td>0.005</td>
<td>0.52</td>
</tr>
<tr>
<td>Bare Brickwork</td>
<td>0.638</td>
<td>72.03</td>
</tr>
<tr>
<td>Painted Brickwork</td>
<td>0.052</td>
<td>5.9</td>
</tr>
<tr>
<td>Stone</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Render</td>
<td>0.233</td>
<td>26.37</td>
</tr>
<tr>
<td>Curtain Walling</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Metal Cladding (painted)</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Metal Handrailing</td>
<td>0.002</td>
<td>0.19</td>
</tr>
<tr>
<td>GRP/Galvanised Steel Cladding</td>
<td>0.001</td>
<td>0.11</td>
</tr>
<tr>
<td>Timber Framing</td>
<td>0.045</td>
<td>5.06</td>
</tr>
<tr>
<td>Timber Cladding</td>
<td>0.014</td>
<td>1.56</td>
</tr>
<tr>
<td>Plastic Board Cladding</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Windows and Doors:</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Stone (frames &amp; mullions)</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Painted Joinery</td>
<td>0.54</td>
<td>6.41</td>
</tr>
<tr>
<td>Alum/Metal/UPVC</td>
<td>0.46</td>
<td>5.46</td>
</tr>
<tr>
<td>Drainage:</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Plastic</td>
<td>0.76</td>
<td>8.06</td>
</tr>
<tr>
<td>Cast Iron</td>
<td>0.24</td>
<td>2.56</td>
</tr>
<tr>
<td><strong>Total:</strong></td>
<td></td>
<td><strong>202.46</strong></td>
</tr>
</tbody>
</table>
The distribution of the five different building types within each grid cell was derived using housing and construction statistics taken from the following sources:

- **Dwellings:** Data were taken from the Central Statistical Office (1991a; 1991b). Where regional figures for the stock of dwellings were unavailable, the number of dwellings was assumed to be proportional to the population of the area, using regional demographic statistics (OPCS, 1991). Data in these sources is given by administrative area. Where boundaries cross the grid squares, the numbers were apportioned between the relevant squares;

- **Schools:** Data were taken from the Central Statistical Office (1991b) divided between grid cells in proportion to the population in each grid square (OPCS, 1991);

- **Shops, offices and industry:** Data were taken from the Commercial and Industrial Floorspace Statistics (CSO, 1991c; Welsh Office, 1990), which provided data at district level. Data for Wales was extrapolated to Scotland based on population data. Wales was used rather than the UK as a whole because, like Scotland, it has a traditionally heavy industrial base, with mainly older industries such as coal, steel and shipbuilding.

During the time of writing this report, an improved inventory of the UK stock at risk has become available (Butlin et al., 1994). This provides a summary of typical building identikits, for a range of modern buildings, for each region of the UK. This reflects the variations in the type of construction and materials used by region and will be used for future analyses.

Some preliminary work was performed on the stock of historic and cultural buildings. There are around 7,500 Grade I buildings in the UK, classified as ‘of exceptional interest’. A catalogue of the UK’s Grade I Buildings by county (Clarke et al., 1980) was used to obtain an estimate of these buildings by grid square. National totals for Grade II* (21,200 buildings) and Grade II (521,000 buildings) were obtained from Historic Buildings Councils Reports (HBCE, 1991; HBCS, 1991; HBCW, 1991). However, we were not able to use these data to study the damages to historic buildings, as it was impossible to develop average identikits for such a varied collection of buildings. Moreover, the assessments necessary to calculate the value of many listed buildings are not available. Further fieldwork is required on the stock of listed buildings, both to estimate the materials inventory and to estimate damage costs, before calculations of damages to these high value buildings can be completed.

**Calculating the UK materials inventory**

The UK materials inventory for each 100 x 100 km grid cell was then calculated by multiplying the average area of each material given by the identikits by the numbers of each building type in each cell. The respective areas for the five building types were added together to obtain the total surface area for each material (stone, brickwork, etc.) in each grid square.

A separate inventory was identified for galvanised steel, which extended the stock at risk to cover the wider use of galvanised steel in general infrastructure. Both 'dipped' products and continuous process products (sheet and strip, wire, etc.) were included. A large number of industrial sectors were analysed, though not all of which were considered to be at significant
risk from atmospheric pollution. For instance, galvanised sheet and strip on automobiles will be corroded by road salts far more than by atmospheric pollution. The sectors identified as being at risk from pollution levels were:

Dipped material:
- Building and construction;
- Street furniture;
- Power sector (electricity pylons);
- Transport infrastructure;
- Agriculture.

Continuous process (sheet and strip):
- Building and construction.

Values were produced from industrial zinc usage in the UK galvanising industry for different industrial sectors, based on a number of statistical sources (in 1990). The average surface area was then calculated based on typical galvanising layers used in industrial sectors (Short, 1994a). For example power transmission pylons have very thick coatings (200 µm) compared to sheet and strip (at around 25 µm). In all sectors, there is a coating limit of around 300 microns. Above this level, there is a tendency for the material to become brittle. It should be noted that more resistant coatings are appearing in the market and that this will affect future damage calculations.

These data were then used to derive a UK stock at risk, shown in Table A5 in the Appendix. For each sector, the proportion of material likely to be exposed to the elements was estimated. This included estimating the proportion of stock which was painted (Short, 1994a). For the ‘best estimate’, this quantity of painted galvanised material was transferred into a separate paint inventory and analysed separately for paint damage and repair. This comprises a significant quantity of material, since painting is important in some industrial sectors, for example in construction, where the majority of dipped products will be painted to fit the local architectural colour scheme. The upper and lower limits on the amount of exposed material give the range of our analysis. The total UK stock, shown in Table A5, was then assigned across the UK into 100 x 100 km grid cells, based on population data.

As pollution occurs on a European range, it was necessary to identify the stock at risk in the rest of Europe. There are no sources of data on building distribution and construction covering the whole of Europe, though individual city inventories do exist. The building material and galvanised infrastructure stock for affected areas outside of the UK was derived from the UK inventory, adjusted for population. For countries at relatively similar levels of economic development this should be a reasonable assumption though building materials will vary significantly across the continent, particularly in different climatic conditions. Over the course of the Project, a number of other building surveys in other European cities have become available. These new inventories provide a more accurate resolution by region and were used for the assessment of the German fossil fuel plants.
Derivation of the German and European stock at risk

The results of two studies were used to quantify the material surfaces of buildings in Germany. Roth et al (1980) identified nine different types of sub-settlements for Germany. As part of this work, a field study assessed the age, the proportion of wall and window area and other attributes of 500 buildings in Freiburg. However, as the study analysed interactions between settlement structures and heating systems, it did not determine the specific materials of the building surfaces. This information was calculated in another study by Hoos et al (1987) who determined the building share for the cities of Dortmund and Köln (Cologne) shown in Table 11.3.

Table 11.3 The Share of Subsettlement Types of Buildings in Köln and Dortmund in 1985.

<table>
<thead>
<tr>
<th>Subsettlement Type</th>
<th>Building share (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Dortmund</td>
</tr>
<tr>
<td>ST1 One and more-family buildings; settlements of low density</td>
<td>4.3</td>
</tr>
<tr>
<td>ST2 Village centres and one-family units; settlements of high density</td>
<td>49.2</td>
</tr>
<tr>
<td>ST3 Row-house-type dwelling settlements</td>
<td>18.9</td>
</tr>
<tr>
<td>ST4 Linear building-type; settlements of medium density</td>
<td>9.6</td>
</tr>
<tr>
<td>ST5 Linear building-type; settlements of high density and high rise bdgs</td>
<td>1.1</td>
</tr>
<tr>
<td>ST6 Development with block residential buildings</td>
<td>10.2</td>
</tr>
<tr>
<td>ST7 City development from the middle of the 19th century</td>
<td>1.7</td>
</tr>
<tr>
<td>ST8 ‘Middle-aged old town’</td>
<td>0</td>
</tr>
<tr>
<td>ST9 Industrial and warehouse building</td>
<td>4.9</td>
</tr>
</tbody>
</table>


Hoos et al undertook a new field survey of 232 buildings to identify the material for the material surfaces for each subsettlement type. This was used to produce a stock at risk for the two cities, shown in Tables 11.A6 and 11.A7 in the Appendix at the end of this chapter. As Tables 11.3, 11.A6/7 show, even the cities of Köln and Dortmund, which are relatively close to each other, have significant differences in material composition in the stock at risk.

To derive the total stock at risk for Germany, statistics on dwelling houses in West Germany were taken from the Gebäude- und Wohnungszählung of 1987. Similar statistics are not currently available for East Germany. For this reason, population data was taken from the REGIOSTAT database provided by the Statistische Bundesamt (German Statistical Office, StaBA). Based on the West German building data and the East German population data, the building identikits of Köln and Dortmund were extrapolated to produce a stock at risk for material surfaces on dwelling houses (for zinc and galvanised steel, sandstone, limestone, other natural stones, and rendering) for the whole of Germany. This assumes that all areas in Germany have the same subsettlement share as Köln and Dortmund.
For other European countries, the best available information was used. The UK material inventory specified above was used for the United Kingdom. For Scandinavia and Eastern Europe, data was taken from a common study (Kucera et al., 1993b; Tolstoy et al., 1990), which derived statistically based inventories of the material quantities on external surfaces of buildings and other constructions for Stockholm (Sweden), Sarpsborg (Norway), and Prague (Czechoslovakia).

In Stockholm and Sarpsborg, all real estates were classified into nine sampling groups, some of which were subdivided into building year classes. Representative samples (randomly selected) of 455 buildings in Stockholm and 191 buildings in Sarpsborg were inspected using special checklists. 60 buildings were surveyed in Prague though a different systematic approach was used which was more detailed. Based on these samples, the total material surfaces were estimated. The material composition on building and construction surfaces in Prague, Sarpsborg, and Stockholm as given by Kucera et al. (1993b) are shown in Table 11.4.

Table 11.4 Building Material Surfaces in Stockholm, Sarpsborg, and Prague (in %).

<table>
<thead>
<tr>
<th>Material</th>
<th>Stockholm</th>
<th>Sarpsborg</th>
<th>Prague</th>
</tr>
</thead>
<tbody>
<tr>
<td>Concrete</td>
<td>14</td>
<td>14</td>
<td>12</td>
</tr>
<tr>
<td>Clay</td>
<td>10</td>
<td>13</td>
<td>9</td>
</tr>
<tr>
<td>Wood</td>
<td>23</td>
<td>30</td>
<td>4</td>
</tr>
<tr>
<td>Rendering</td>
<td>14</td>
<td>9</td>
<td>23</td>
</tr>
<tr>
<td>Metals</td>
<td>19</td>
<td>10</td>
<td>40</td>
</tr>
<tr>
<td>Glass</td>
<td>5</td>
<td>5</td>
<td>5</td>
</tr>
<tr>
<td>Bitumen felt</td>
<td>9</td>
<td>10</td>
<td>5</td>
</tr>
<tr>
<td>Others</td>
<td>6</td>
<td>9</td>
<td>2</td>
</tr>
</tbody>
</table>

Source: Kucera et al. (1993b).

The total material surface per inhabitant for the three cities surveyed were 83 m²/inhabitant in Prague and 165 and 132 m²/inhabitant in Sarpsborg and Stockholm respectively (Kucera et al., 1993b). It was assumed that 20% of the metal was zinc and 30% was galvanised steel. These values were then used with the other data to build up a regional picture for Europe.

For Eastern Europe, the building identikit of Prague has been used; for Scandinavia, the identikits of Sarpsborg and Stockholm. The extrapolations in both cases were based on population data taken from the EUROGRID database. For all the remaining Western and Southern European countries, the building identikits of Köln and Dortmund were extrapolated, again based on the population data compiled in the EUROGRID database. The resulting total material surfaces are summarised by region in the Appendix in Table 11.A8. The inventory was then calculated for each of the large EUROGRID cells (100 x 100 km) across Europe, based on the broad regional categorisations for material composition/person.
During the time of writing this report, an improved inventory of the stock at risk has been compiled for Northern and Eastern Europe based on a more detailed analysis of Kucera et al., (1993b), this incorporates material types for different purposes and will be used for future analyses.

With the exception of the UK and West German data, the extrapolation of building identikits to Europe is based on population. This method involves a number of assumptions. For example, the use of the data above assumes that in every Western European village, the percentage of skyscrapers is as high as in Dortmund or Köln. However, in the absence of more detailed data, the use of population statistics is a reasonable method to produce our stock at risk. Alternative approaches could improve future estimates. The use of 'educated guesses' could improve the extrapolation, for example by adapting stone quantities in rural areas, though there is no way such approaches can be verified. A more accurate but time-consuming approach was developed by Roth et al (1980), who categorised regions according to characteristic population and housing distributions into four regional area types.

- City regions:
  1. Inner cities;
  2. Border areas.
- Surrounding regions:
  3. Middle-sized towns outside of city regions (>10 000 inhabitants, >250 inhabitants/km$^2$);
  4. Rural areas.

Each of these regional areas consisted of settlements (as towns or villages), which in turn consisted of subsettlements (see subsettlement list in Table 11.3). These subsettlements have a characteristic image in topographic maps with resolutions greater than 1:50 000.

Map images from all West German communities (chosen as a representative 1% random sample) were analysed and the area of each subsettlement type were determined, based on the regional area type. A correlation analysis was then carried out between these subsettlement type areas and attributes registered in the 'Gemeindestatistik' (community statistics) of 1970. Attributes with a coefficient of determination $R$ better than 0.95 were used to derive the regression equations. Examples of these attributes were:

- Number of non-farming houses with one dwelling;
- Number of farming houses;
- Number of dwellings constructed up to 1900;
- Employees in trade and traffic.

Interestingly, the number of inhabitants was not included as its coefficient of determination was too low.

The application of this model would provide more reliable data than a simple extrapolation of the Dortmund and Köln data to the whole of Germany (and in turn to Europe). However, there were a number of reasons which did not allow this, primarily because of changes in the reporting of statistics and regional reforms.
A repetition of the map survey and correlation analysis would be necessary but it is too time-consuming for this project. It is possible that in the near future, such distributions will be relatively easy to calculate using Geographical Information Systems. At present, we do not consider that the error introduced from using population data is too great, given the uncertainty in other parts of the analysis, though this area does warrant future investigation.

**Meteorological, atmospheric and pollution data**

The data required to complete the description of the reference environment are the meteorological conditions which affect damage. Of these, the most important are precipitation and humidity.

For the UK studies, the following data were used:
- Precipitation averages (UKMO, 1977) were used to calculate average rainfall values for each grid square in the UK.
- Average relative humidity figures were taken from UKMO (1970).
- Estimated percentage of time that humidity exceeds critical levels of 80%, 85% and 90% were taken from UKMO (1975).
- Data on UK chloride concentrations were derived from the Harwell Trajectory Model.
- Precipitation outside the UK was uniformly assumed to be 0.6 m/year (this is within a factor of 2 of all major centres of population in Europe);
- UK background ozone levels were assumed to be 40 µg/m³ (Kucera, 1994);

For the UK, good data is available for relative humidity. Data for ozone levels are also now available by grid cell and will be used in future assessments. However, for the rest of Europe, good data is not available on both these parameters. To date, the analysis of UK plant emissions has been constrained to incremental acidity outside of the UK (i.e. direct SO₂ increments were only calculated for the UK). For this reason, uncertainties with respect to O₃ levels or time of wetness do not affect our estimates of incremental damage in Europe and so average background humidity, chloride deposition and ozone levels from the UK were applied to the rest of Europe.

For the German implementation, the estimated percentages of time that relative humidity exceeds 85% for several measurement stations were taken from reports of the Deutsche Wetterdienst (German meteorological service) (Cappel and Kalb, 1976; Kalb and Schmidt, 1977; Schäfer, 1982; Bätjer and Heinemann, 1983; and Höschele and Kalb, 1988) and used as average for Germany. For the other climate parameters in other regions, including average O₃ concentrations, the data measured in the UN ECE ICP study (Kucera, 1994) were averaged for each region and used. Because of the sensitivity of some materials to ozone (acting synergistically with SO₂), a lack of good spatially resolved data on background level introduces an important source of uncertainty into the analysis. More accurate data from across Europe should be fed into the study to improve future estimates of damage.

For the implementation of plant emissions in both countries, atmospheric models were used to look at wet and dry pollution deposition rates. The Harwell Trajectory Model (and the
Windrose Trajectory Model) described in Chapter 3 were used to model background pollution levels, and the levels with the addition of the power plants. The output were used to quantify damages using dose-response functions linking pollution levels to surface corrosion.

11.4 Identification of Dose-Response Functions

Dose-response functions, showing the rate of loss of material for the building materials considered to be at risk, were identified by literature review. Considerable relevant literature now exists both in the USA and Europe.

Estimates of pollution related damage rely on field studies on real buildings or upon studies on idealised test materials either in the field or laboratory studies. There are obvious problems in relying too heavily on test materials because real building materials are very varied in type and in the atmospheric conditions they face. Also, the mechanisms as well as the rates of damage can vary with ambient conditions. On the other hand, surveys of actual building materials do not allow the same flexibility in control of individual pollutants in a multi-pollutant environment. A combination of controlled studies and field measurement is therefore required to give reliable dose-response data.

All attempts to derive dose-response relationships require some assumptions to be made about the appropriate functional form. For example, responses that are linear with respect to time and to the concentration of individual atmospheric pollutants are common assumptions. In some cases the basis for these assumptions may be very weak as the empirical data sets are often insufficient to test the hypothesis. In these cases, the functional form needs to be examined on theoretical grounds with respect to models of damage mechanisms to ensure that the functions are at least reasonable.

The use of measured data to derive (via regression analysis) predictive equations with empirically fitted coefficients does mean the coefficients only strictly apply to the experimental environment. Caution is therefore needed when comparing functions obtained from data taken from widely different environments. For the purposes of this study it is assumed that dose-response functions are geographically transferable. However, where possible, we have preferentially used functions that are specific to the region of assessment. Finally, when applying functions to the material components of actual buildings it should be appreciated that the micro-climate at different locations around a structure will not be the same, for instance from sheltered areas.

The functions we have used are derived from studies in several countries. In all cases, we have tried to recommend the use of a range of values from the best of the available functions. At the time of writing, no one set of functions can be regarded as definitive. Evidence presented in this Chapter demonstrates that these relationships can change significantly with time, as a result of variation in the relative importance of the mechanisms that promote, and conversely protect against, erosion and corrosion for any species.

In general, we have concentrated on three studies for our assessments of building damages: Lipfert (1987; 1989), the UK National Materials Exposure Programme (Butlin et al, 1992a;
The ICP functions are preferred and we recommend their use in future studies. A comparison of the basis for the exposure-response relationships for natural stone and metals from these studies is shown in Table 11.5.

### Table 11.5 Comparison of the Dose-Response Functions for Material Damage Assessment.

<table>
<thead>
<tr>
<th>Exposure time</th>
<th>Kucera</th>
<th>Butlin</th>
<th>Lipfert</th>
</tr>
</thead>
<tbody>
<tr>
<td>4 years</td>
<td>2 years</td>
<td>-</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Experimental technique</th>
<th>Uniform</th>
<th>Uniform</th>
<th>Meta analysis</th>
</tr>
</thead>
</table>

<table>
<thead>
<tr>
<th>Region of measurement</th>
<th>Europe</th>
<th>UK</th>
<th>-</th>
</tr>
</thead>
</table>

<table>
<thead>
<tr>
<th>Derivation of relationships</th>
<th>Stepwise linear regression</th>
<th>Linear regression</th>
<th>Theoretical</th>
</tr>
</thead>
</table>

The UK National Materials Exposure Programme was commissioned by the UK DoE. The programme consists of 29 sites across the UK and includes a range of materials including aluminium, copper, Portland limestone, White Mansfield dolomitic sandstone, Monks Park limestone, and mild, painted and galvanised steel. The samples were exposed in 1987 and were retrieved after 1, 2 and 4 years of exposure. Four of the sites also form part of the ICP programme. Since the initial implementations, the four year results from this study have become available and will be used for future analyses.

The International Cooperative Programme (ICP) as part of the UN ECE study (Kucera, 1994) is particularly noteworthy because data is being collected at no fewer than 36 test sites throughout Europe (in the Czech Republic, Estonia, Finland, Germany, Italy, the Netherlands, Norway, Portugal, Sweden, Spain, Russia and the UK) and three sites in Canada and the USA. These sites clearly cover a broad geographical region, with substantial variation in climate and pollution exposure regime. The aim of this programme is to perform a quantitative evaluation of the effects of sulphur pollutants in combination with nitrogen oxides and other pollutants, as well as climatic parameters, on the atmospheric corrosion of important materials in a wide geographical zone of Europe and North America. A uniform experimental protocol is being used. The exposure programme started at all test sites in September 1987 and was due to last for 8 years.

The ICP work is not yet complete, and hence finalised relationships are not available. Comparison of the relationships derived after 2 and 4 years of exposure (Kucera, 1994) reveals variation in both magnitude of predicted effect and functional form with time. The study has produced functions for steel, weathering steel, aluminium, copper, bronze, nickel and silver (electric contacts), limestone, sandstone and paint coatings. The effects on glass and polymers are also under analysis although no functions have been derived as yet. The study provides functions for both sheltered and unsheltered material.

The following sections describe background information on each material and list the dose-response functions we have considered. A summary of the functions we recommend is included at the end of this section. The following key applies to all equations given:
Impacts on Materials

\[
\begin{align*}
\text{ER} & = \text{erosion rate (\(\mu\text{m/year}\))} \\
\text{P} & = \text{precipitation rate (m/year)} \\
\text{SO}_2 & = \text{sulphur dioxide concentration (\(\mu\text{g/m}^3\))} \\
\text{O}_3 & = \text{ozone concentration (\(\mu\text{g/m}^3\))} \\
\text{H}^+ & = \text{acidity (meq/m}^2/{\text{year}}) \\
\text{RH} & = \text{average relative humidity, } \% \\
\text{f}_1 & = 1 - \exp[-0.121.\text{RH}/(100-\text{RH})] \\
\text{f}_2 & = \text{fraction of time relative humidity exceeds 85\%} \\
\text{f}_3 & = \text{fraction of time relative humidity exceeds 80\%} \\
\text{TOW} & = \text{fraction of time relative humidity exceeds 80\% and temperature } > 0^\circ\text{C} \\
\text{ML} & = \text{mass loss (g/m}^2\) after 4 years \\
\text{MI} & = \text{mass increase (g/m}^2\) after 4 years \\
\text{CD} & = \text{spread of damage from cut after 4 years, mm/year} \\
\text{Cl}^{-} & = \text{chloride deposition rate in mg/m}^2/{\text{day}} \\
\text{Cl}_{(p)}^{-} & = \text{chloride concentration in precipitation (mg/l)} \\
\text{D} & = \text{dust concentration in mg/m}^2/{\text{day}}
\end{align*}
\]

It should be noted that there is considerable variation in the use of above terms in the original equations. For example, precipitation is frequently referred to in mm and m, the time of wetness term is often differently defined, and invariably the H\(^+\) term is in different units. Throughout the following sections, we have tried to standardise all the functions for ease of comparison, though the reader should refer back to the original papers for the original form and discussion of the functions.

In all the ICP functions, the original H\(^+\) concentration term (in mg/l) has been replaced by an acidity term using the conversion:

\[
P\cdot\text{H}^+(\text{mg/l}) = 0.001\cdot\text{H}^+(\text{acidity in meq/m}^2/{\text{year}})
\]

In addition, as these equations are written in units of mass loss, rather than erosion rate, to convert mass loss for stone and zinc into an erosion rate in terms of material thickness, we have assumed respective densities of 2.0 and 7.14 tonnes/m\(^3\).

11.4.1 Natural stone

The types of stone commonly used for buildings and monuments include granite, sandstone, limestone, marble and slate. The durability of these stones are determined by their individual composition and porosity. Granites are composed mainly of silica as quartz, and have low porosities and generally good durability. Limestones are predominantly calcium carbonate (calcite) and are much more porous and susceptible to attack. Sandstones are composed of quartz grains bonded together by siliceous or calcareous ‘cement’, the latter with a durability similar to limestone. Finally, marble has a dense crystalline structures and consists of almost pure calcite. Reviews (Harter, 1986; Lipfert, 1987; Lipfert, 1989; UKBERG, 1990; NAPAP, 1990) have concluded that acid deposition damage to siliceous stones is negligible, and therefore attention is confined here to calcareous stones, i.e. limestone, marble and calcareous sandstones. These are extensively used as building materials within Europe.
Impacts on Materials

The full impact pathway for stone was shown in Figure 11.1. We do not attempt to provide a complete picture here of all the reactions and parameters important in stone erosion, merely a summary of the important points relevant to our analysis. A fuller discussion of the effects of erosion on stone and on the dose-response functions we use can be found elsewhere (Cooke and Gibb, 1994; Short, 1994b).

The weathering of stone occurs naturally, primarily because of carbon dioxide present in the atmosphere. CO₂ dissolves in rain-water, producing an acidic solution. When in contact with calcareous stone, this acidic rainwater slowly reacts with the calcium carbonate in the stone to form calcium bicarbonate; this is soluble and is readily washed away. Any bicarbonate solution that remains on the surface on in pores and subsequently evaporates will re-precipitate calcium carbonate on the surface (Cooke and Gibbs, 1994). The amount of calcite removed is a function of CO₂ concentration, temperature and the physical characteristics of the stone, such as porosity. The overall effect is a long-term thinning of the stone. Water running off horizontal surfaces and down vertical surfaces is not usually uniform and this gives rise to dirt streaks and erratic etching. In addition to this natural chemical attack there are many other natural damage mechanisms affecting stone. These include stresses from water freezing and salt crystallisation cycles in the stone, which can lead to blistering and exfoliation. In addition, particles and rainfall can cause surface abrasion.

In the presence of SO₂ a much faster chemical attack occurs. This can occur via additional dissolution from rain acidity or from attack by dry deposition of pollutants. The presence of SO₂ decreases the pH of rain, causing reactions to accelerate and the formation of calcium sulphate, which is more soluble than calcium carbonate. For low porosity stones, where the surface is frequently washed by rain, then generally deterioration products do not accumulate and are continuously washed away in run off. The surface looks unaltered and very clean, but a very thin layer of stone is removed and the surface consists of re-crystallised calcite. In more porous stones, inward diffusion of acidic solution may occur resulting in the re-crystallisation of calcium sulphate within the pores.

Dry deposition of SO₂ also leads to chemical attack of the calcium carbonate. In an urban environment, the dry deposition to the stone surface may be more than a factor of ten greater than wet deposition of ions. Dry deposited ions can be later activated by only a small amount of water. The addition of water to these often sheltered zones forms an aggressive solution which is much more concentrated than from natural processes or from additional dissolution owing to rain acidity. The solution dissolves and progressively transforms the carbonate surface into the crystalline product gypsum (hydrated calcium sulphate). The exact formation of the solution involves a complex interaction mechanism though a crucial stage is the oxidation of SO₂ to sulphate. The mechanism depends very much on the prevailing atmospheric conditions, and both particulate solids and ozone appear to catalyse the process. Evidence now also suggests that NOₓ may also enhance these reactions involving SO₂.

For areas of stone which are directly exposed to rainfall (or run-off), then during wet periods the calcium sulphate will be washed away. In sheltered areas, the calcium sulphate accumulates on the surface within surface pores of the stone. During drier periods, water
evaporates and a gypsum crust is formed. Carbonaceous particulate solids are also incorporated into the crust making it black. Eventually, depletion of calcite behind the crust leaves a weakened layer. During alternate wet/dry cycles the crust expands/contracts, internal stresses build up, and subsequently the crust separates from its substrate to produce spalling (exfoliation) often with the loss of underlying material. Further attack by either acid rain or freeze/thaw cycles on the weakened stone structure may accelerate the cycles of damage.

Identification and use of stone dose-response functions

The deterioration of stone can be broken down into three processes, categorised as:

1. **Stage I** (short term). This involves simple dissolution of calcium carbonate. It includes (i) normal dissolution of calcite in rain from CO₂, (ii) acceleration owing to rain acidity as a result of air pollution, (iii) attack by dry deposition of gaseous pollutants especially SO₂.

2. **Stage II** (medium term). This includes the dissolution of calcium carbonate plus the fall-out of less soluble granular particles within the matrix e.g. for calcareous sandstones, the removal of small amounts of the calcium carbonate matrix may loosen a considerable number of sand grains, leading to more severe surface erosion.

3. **Stage III** (long term). In sheltered areas where calcium sulphate is not intermittently washed away, there is a non steady build-up of salts. This results in the formation of a crust which may be followed by exfoliation.

Stage I processes can be characterised simply by stone recession, which is easy to measure. The quantification of damages for stage II and III processes are less easy to define and fewer dose-response functions, if any, are available. For this reason, attention in our study has focused on analysing stage I processes, acknowledging that this underestimates later, more severe damage.

For stage I processes in calcareous stone, the mass loss in exposed material depends on dry deposition of SO₂, the time of wetness and on the load of acidity from rain. A wealth of studies have characterised the effect of these parameters on stone erosion and many dose-response functions exist. We have applied functions taken from the three main studies we consider; namely the work of Lipfert, Butlin and Kucera.

Lipfert (1989) produced a theoretical universal damage function for generic stone loss. A meta-analysis of data from both sides of the Atlantic provided similar but not equal relationships. For these data, Lipfert found regression coefficients close to the theoretical value for the wet acid deposition and dry deposition terms. The natural solubility term had a regression coefficient less than a third of the theoretical value (Webb *et al*., 1992).

Lipfert - natural stone:

\[
ER = 18.8 \cdot P + 0.052 \cdot SO_2 + 0.016 \cdot H^+ 
\]  

[1]

The function (equation [1]) is strongly precipitation dependent and indicates a rather lower acid erosion rate than determined in some earlier studies. An alternative dose-response function based on preliminary results from the UK NMEP programme for exposed Portland limestone (Butlin, 1992a). The materials and environmental conditions for this function are
characteristic of the UK. The original equation was for \% weight loss over two years; adjusting corrosion in µm, the function for annual erosion rate is shown in equation [2].

**Butlin - Portland limestone:**

\[
ER = 2.56 + 5.1\cdot P + 0.32\cdot SO_2 + 0.083\cdot H^+ \quad [2]
\]

The most recent dose-response functions for limestone samples are obtained from the four year exposures within the International Cooperative Programme (Kucera, 1994). A stepwise linear regression analysis of the measurement data was performed. The important terms among the measured factors were the \(SO_2\) concentration, the wet acid deposition, the time of wetness and the rainfall. The dose-response function derived for the limestone mass loss over the four year exposure is shown in equation [3].

**ICP - unsheltered limestone (4 years):**

\[
ML = 8.6 + 1.49\cdot TOW\cdot SO_2 + 0.097\cdot H^+ \quad [3]
\]

In order to analyse stage II effects, it is necessary to take into account the loss of inert particles as the matrix is dissolved away. No functions for stage II impacts on natural stone exist as such, though functions for calcareous sandstones are available as determined by linear regression analysis (Butlin, 1992a; Kucera, 1994). Further analysis of the data is, however, required. The Butlin function (1992a) for White Mansfield sandstone is shown in equation [4] adjusted from the original \% weight loss over two years into annual erosion rate. The negative \(NO_2\) term may be an artefact from correlation with another variable or because the \(NO_2\) term has a significant negative relationship. The results of the 4 year study should resolve this. The ICP function (Kucera, 1994) for sandstone is shown in equation [5], showing average mass loss over the total four year exposure.

**Butlin - sandstone:**

\[
ER = 11.8 + 1.3\cdot P + 0.54\cdot SO_2 + 0.13\cdot H^+ - 0.29\cdot NO_2 \quad [4]
\]

**ICP - unsheltered sandstone (4 years):**

\[
ML = 7.3 + 1.56\cdot TOW\cdot SO_2 + 0.12\cdot H^+ \quad [5]
\]

The prediction of stage III mechanisms is complicated. The mass of samples initially increases through the formation of a crust of gypsum on the stone surface. This layer eventually flakes off, often removing some underlying material with it (Cooke and Gibbs, 1994). No functions exist for stage III processes - the time scales required to produce thick crusts is very long and when crusts exfoliate the action is very damaging. Such a process is very difficult to characterise. Functions do exist for the early mass increase on samples sheltered from the rain from the ICP study (Kucera, 1994). However, these functions (equations [6] and [7]) do not quantify later damages, such as exfoliation.

**ICP - sheltered limestone (4 years):**

\[
MI = 0.59 + 0.20\cdot TOW\cdot SO_2 \quad [6]
\]

**ICP - sheltered sandstone (4 years):**

\[
MI = 0.71 + 0.22\cdot TOW\cdot SO_2 \quad [7]
\]
The mechanisms of crust formation would imply that kinetics should be essentially parabolic and rates would depend on diffusion of solution in and out of the growing layer. Equations [6] and [7] are linear which suggests that the increased layers are very thin and that parabolic rates have not manifested themselves in the time period. The results of the eight year ICP samples should help to resolve the lack of functions for predicting these later processes.

11.4.2 Brickwork, mortar and rendering

The current opinion is that brick, which is a calcium-aluminium silicate ceramic, is unaffected by sulphur dioxide attack. However, although brick itself is relatively inert to acid damage, the mortar component of brickwork is not.

Mortar consists of sand, calcium hydroxide and other carbonate phases. The primary agent of mortar erosion is acid attack on the calcareous cement binder (UKBERG, 1990; Lipfert, 1987). As no specific dose-response functions for mortar exist, preliminary estimates have to be made on the basis of theoretical comparisons with calcareous stone. SO$_2$ reacts with the calcareous component of common lime or Portland cement mortars in a similar way to natural stones and Portland cement. These mortars have a relatively high porosity and are generally less durable than fired clay products or concrete. However, the extent of reaction will depend very much on the properties of the particular mortar; thus lime and weak Portland cement mortars will be more susceptible than strong Portland cement mortars with well graded sand.

As with the stage II damages for calcareous sandstone above, it is the calcite which is the cementing agent in mortar and rendering that binds the sand aggregate together and so erosion rate is determined by this cement erosion. As the calcite dissolves, the entire mixture crumbles and is lost, so that a multiplication effect results.

The proportion of calcareous material in the original matrix should give an indication of the magnitude of damages. The initial estimates in the ExternE Project assumed mortar (and rendering) were typically one third calcite and hence the dose-response functions used for mortar were simply three times that used for natural stone. However, diversity of mortar types suggest that there is considerable uncertainty in this type of estimate. There is no sound basis for this approach and since other dose-response functions have become available it seems to be more appropriate to consider these. The mechanism of deterioration is considered to be more like that of sandstone (Short, 1994b) and therefore equations [4] and [5] above should be used, with no adjustment. However, it should be noted that this will probably underestimate damages, since mortars are likely to be more porous than sandstones.

Mortar may also be susceptible to efflorescence. This occurs when soluble salts and water migrate through the porous matrix of mortar, the salts being deposited at the surface after the water has evaporated. Repeated dissolution and re-crystallisation in this manner may lead to crumbling of the mortar or disruption of brickwork. The salts tend to concentrate in the more porous material. The capability of a given masonry structure to resist deterioration will be directly related to its resistance to water penetration which is the main single agent responsible for the decay process by salt crystallisation. Whilst acid rain may exacerbate this
effect, sulphate from other sources, e.g. sodium sulphate in brick, and magnesium salts in the mortar or ground water are probably more important (Short, 1994b). It should be noted that this problem may be more important in very old masonry.

11.4.3 Concrete

Portland cement, the major binding agent in most concrete, is an alkaline material which is susceptible to acid attack. Potential impacts to concrete include soiling/discolouration, surface erosion, spalling and enhanced corrosion of embedded steel. However, for all these impacts (with the exception of surface erosion) damages are more likely to occur as a result of natural carbonation and ingress of chloride ions, rather than interaction with pollutants such as SO$_2$.

The main factor influencing the durability of concrete is the corrosion of embedded steel reinforcement. This is generally present as steel reinforcing bars. In new concrete, these bars are protected from acid corrosion by the alkaline characteristics of the cementitious component of the concrete. Exposure to air and rain over a period of years neutralises this, causing the pH of the cement paste to fall and leaving the steel open to corrosion (UKBERG, 1990). Essentially this neutralisation involves reaction between CO$_2$ and SO$_2$ in the atmosphere with the calcium hydroxide of the cement pastes. From a thermodynamic point of view, sulphation rather than carbonation reactions are favoured. In practice, however, carbonation is much more important because of the significantly higher carbon dioxide concentrations in the atmosphere and this becomes the dominant process.

Therefore, provided the concrete layer covering the steel is thick enough and the concrete is of good quality, then acidic pollutants only etch the surface. As surface erosion does not affect structural integrity it has no impact on concrete durability. As no costs arise from incremental pollutant concentrations, we have not included pollution damage to concrete in our assessment.

There is, however, some evidence that when the concrete layer is too thin or badly prepared, atmospheric pollutants may have a role in accelerating damage. In badly prepared concrete, when carbonation has already led to steel corrosion, cracks will develop rendering the system more accessible to attack by SO$_2$, as the corrosion products of steel occupy a greater volume than the steel (Webster and Kukacka, 1986). It is also possible that SO$_2$ reacts instantaneously with the calcium hydroxide phase in cement to form gypsum and ettringite. This can lead to the build-up of internal stresses and the breakdown of the cement properties, causing crumbling of the concrete (Pye, 1986). No quantitative information exists for these processes, and the uncertainty surrounding the effects of SO$_2$ on concrete is still high.

11.4.4 Paint and polymeric materials

Damages to paint and polymeric materials can occur from acidic deposition and from photochemical oxidants, particularly ozone. The acidic deposition damage pathway for paint and other polymeric materials was shown in Figure 11.3. Potential impacts include loss of
gloss and soiling, erosion of polymer surfaces, loss of paint adhesion from a variety of substrates, interaction with sensitive pigments and fillers such as calcium carbonate, and contamination of substrate prior to painting leading to premature failure and mechanical property deterioration such as embrittlement and cracking particularly of elastomeric materials.

Although ozone is known to damage some polymeric materials such as paints, plastics and rubbers to date we have not considered such damages. Further development is needed on modelling fossil fuel stack emissions on O$_3$ levels and on identifying suitable dose-response functions linking O$_3$ to material damage. For this reason, we concentrate on acidic deposition effects. Nonetheless, the damages from ozone effects on materials are seen as a priority for future assessments.

Paints are complex mixtures of polymers, pigments, and extenders. They also include a number of other additives used to improve properties such as the adherence to the substrate. As well paint composition, the exact damage mechanisms to paint work will depend upon the type of substrate, the surface preparation before painting, the quality of application and the ambient conditions.

SO$_2$ is thought to be important in the degeneration of paint systems (paint and paint substrate) exposed in the real environment. The rapid diffusion of pollutants such as SO$_2$ through polymeric paint films to the substrate may be important (Williams, 1986). This is followed by reaction at the paint/substrate interface, resulting in loss of adhesion of the paint, and/or deterioration of the substrate.

Where surface preparation is poor cracking and/or delamination of paint work can occur, allowing direct damage of the substrate by the atmosphere. In these cases, the critical factor is the damage to the substrate or to the interface between substrate and paint, not damage to the paint.

The direct reaction of acidic pollutants with the pigments and fillers present in paint can accelerate erosion. The most serious impact studied involves the influence of SO$_2$ on paints with calcium carbonate fillers. In this case the mechanism of interaction between SO$_2$ and the calcium carbonate were thought to be similar to those for calcareous stones. Nitrogen oxides have only a minor effect on paints (Spence et al, 1975; Haynie and Spence, 1984). The role of NO$_x$ in the deterioration of flawed paint systems has yet to be determined.

Identification and use of paint dose-response functions

The main pollution related impact for which dose-response functions have been derived is the erosion of paint work (NAPAP, 1990). Care must be applied to ensure that dose-response functions are applied to the grade of paint for which they were derived. For example, one function derived from US data which is often used (Haynie et al, 1976) applies to oil based paints that are now rarely used for house painting. The only European dose-response function (Glomsrød and Rosland, 1988) relies to some extent on US data and does not distinguish between different categories of paint.
The most extensive review is from the USA (Haynie, 1986). This identifies a 10-fold difference in acid resistance between carbonate and silicate based paints. In the US, silicate based paints predominate in the house paint market (NAPAP, 1990). However, discussions with a leading European manufacturer confirm that the overwhelming majority of UK house paints contain calcium carbonate and that this component is particularly important in the cheaper grades of paints (ICI, 1992). It is used as a low cost extender to the pigment component of the paint, the pigment typically being 25%-40% of the paint by weight. In the UK 9% by weight of all paint is calcium carbonate (PRA, 1992), confirming that paints with carbonate extenders have a large market share.

The dose-response function for carbonate based paints (equation [8], Haynie, 1986) is therefore appropriate for UK house paints, in which $t_c$ = the critical thickness loss, which is about 20 µm for a typical application:

\[ \Delta ER/t_c = 0.01 \cdot P \cdot 8.7 \cdot (10^{-pH - 10^{-5.2}} + 0.006 \cdot SO_2 \cdot f_1) \]  [8]

According to German manufacturers, carbonate based paints have been replaced by acrylic and epoxy resins based paint over the last decade. Carbonate based paints are no longer used. Thus, in mainland Europe, Haynie’s function for silicate paints (equation [9]) is also considered to provide a range for the assessment.

\[ \Delta ER/t_c = 0.01 \cdot P \cdot 1.35 \cdot (10^{-pH - 10^{-5.2}} + 0.00097 \cdot SO_2 \cdot f_1) \]  [9]

The Haynie functions were derived from laboratory prepared samples for unflawed paint on inert substrates. Therefore, they do not take into account corrosion of the substrate and substrate/paint interactions that together provide an alternative mechanism for paint decay.

However, paint failures frequently occur before linear erosion of the film, as a result of blistering, blooming, cracking and chalking. The extent to which pollution is implicated in these processes is not known, though there is good reason to suspect some association. Current research in Sweden uses blistering rates to assess paint-system decay (Lampe and Saarnak, 1986). The dose-response information for real life exposure of painted steel in the external environment is expressed in terms of lifetime of the paint coating.

Some preliminary data are available from the ICP study. Samples (from 4-year exposure times) have shown damage from cuts, chalking, dirt and fungi on paint surfaces. A correlation analysis has shown that for the spread of defects on scratched steel panels, the $SO_2$ and $O_3$ concentrations are the most significant parameters (Kucera, 1994). The function is shown in equation [10], and is the only equation obtained for painted surfaces so far in the study. The function gives a value (in mm) for the spread of damage on panels after 4 years. However, it is difficult to see how this function could be implemented within the present study, because of the difficulty in applying to our inventories. It is hoped that further results will appear with the results of the ICP eight year samples.

\[ CD = -6.1 + 0.18 \cdot SO_2 + 0.18 \cdot O_3 \]  [10]

ICP - paint (4 years):
11.4.5 Metals

Atmospheric corrosion of metals is generally an electrochemical process. It takes place in corrosion cells with anodes and cathodes. These cells can only operate in the presence of an electrolyte, so atmospheric corrosion only occurs when the surface is wet. The rate of metal corrosion is determined by the interaction of different climatic parameters, the most important of which are humidity, precipitation, temperature and levels of atmospheric pollutants. Of the atmospheric pollutants, SO$_2$ causes most damage, though in coastal regions chlorides also play a significant role. The role of NO$_x$ and ozone in the corrosion of metals is uncertain, though recent evidence (Kucera, 1994) shows that ozone may be important in accelerating some reactions.

Although dose-response functions exist for many metals, we have confined our analysis to those for which we have good inventory data for, i.e. steel, galvanised steel (zinc) and aluminium. Other metals would be important if we were to extend our material inventories, for example copper present in historic buildings and monuments.

Steel is well known to suffer corrosion even under normal ambient conditions. The rates of corrosion are observed to increase both in polluted industrial environments and in salty conditions typical of coastal areas (UKBERG, 1990). However, steel is virtually always used coated with paint (when not galvanised) and we have considered corrosion of steel separately in this analysis. The stock of steel in our inventories has therefore been transferred to the paint stock at risk.

11.4.7 Zinc and galvanised steel

Zinc is not an important construction material itself, but is extensively used as a coating for steel, known as galvanised steel. Zinc has a lower corrosion rate and is corroded in preference to the substrate, thereby acting as a protective coating.

In the absence of pollutants, the initial reaction of zinc with the atmosphere results in the formation of zinc oxide and zinc hydroxide which in turn are converted to the relatively insoluble zinc carbonate. The composition of the resulting layer is quite complex consisting of oxide, hydroxide and carbonate, in proportions depending on actual exposure conditions.

Such films are not completely protective since zinc hydroxide and carbonate (or bicarbonate) are slowly washed away, but they do tend to inhibit continuation of the corrosion process. The wet or dry deposition of acidic species results in a more rapid dissolution of the zinc and/or zinc corrosion products with the additional formation of zinc sulphate. As the reaction proceeds, the pH of the surface solution rises and eventually a basic salt, usually zinc carbonate, precipitates. Again, these films are not completely protective as the zinc sulphate and carbonate (or bicarbonate) are washed away. Film formation and dissolution are a complex function of pH, time of wetness, degree of exposure etc. and the overall rate of corrosion remains approximately constant.
The effects of pollution on zinc corrosion is dominated by dry deposition of SO$_2$; the acidity of rain has a secondary role to play. Recent laboratory studies have shown that NO$_2$ and ozone accelerate the corrosion of zinc in humid atmospheres containing SO$_2$ (Svensson and Johansson, 1993). It was thought that ozone oxidises SO$_2$ to sulphate resulting in an increased rate to SO$_2$ deposition, whilst NO$_2$ catalyses the oxidation process. The effects of ozone on corrosion rate has also been observed in the ICP studies (Kucera, 1994). Chlorides do increase the corrosiveness of rain, but can be ignored except in coastal regions.

**Identification and use of zinc dose-response functions**

Despite a large number of studies of zinc corrosion over many years, there still remains some uncertainty about the form of the dose-response function. One review (UKBERG, 1989) identifies 10 different functions that assume time linearity, consistent with the expectation that the products of corrosion are soluble and therefore non-protective. However, other reviews (Harter, 1986 and NAPAP, 1990) identify a mixture of linear and non-linear functions. It is postulated (Lipfert, 1987) that the formation of zinc carbonate could result in non-linearity.

It is clear that many uncertainties remain in this field. Not least of these in the penetration of more corrosion resistant zinc coatings onto the market. For this study, we have used the same three studies to provide dose-response functions. The first, shown in equation [11], is the result of a meta-analysis by Lipfert (1987) based on theoretical principles and eight test programmes, comprising 72 different test sites, from a number of different research projects in various countries. It is presumed that the individual test programmes used different experimental techniques over different periods of time. The study proposes a complex dose-response function with kinetics intermediate between linear and parabolic. There is a strong dependence on time of wetness and an additional term is included to represent the removal of a protective film by wet deposition. The function also has a time function, which complicates use within our study.

**Lipfert - unsheltered zinc (annual loss):**

\[
ML = [t^{0.78} + 0.46\log_e(H^+)]\times[4.24+0.55f_2\cdot SO_2+0.029\cdot Cl^-+0.029\cdot H^+]
\]  \[11\]

The second function we have considered is from Butlin et al (1992a), shown in equation [12]), the result of the first year exposures in the NMEP study. This is a simpler function, but it has the disadvantage of only attributing damage to SO$_2$. As part of our sensitivity analysis, we have considered this function in the assessment of UK damages, though the function has not been included in the best estimates.

**Butlin - unsheltered zinc (one year):**

\[
ER = 1.38 + 0.038\cdot SO_2 + 0.48P
\]  \[12\]

An alternative function has been derived from the ICP UN ECE study (Kucera, 1994). This function is dependent on the parameters for ozone and the time of wetness; both of which are not well characterised across Europe. This introduces some uncertainty into the analysis. Other factors include the SO$_2$ concentration, acid concentration in precipitation and rainfall.
The functions for unsheltered and sheltered corrosion after 4 year exposures are shown in equations [13] and [14].

ICP - unsheltered zinc (4 years):
\[ ML = 14.5 + 0.043 \cdot TOW \cdot SO_2 \cdot O_3 + 0.08 \cdot H^+ \] [13]

ICP - sheltered zinc (4 years):
\[ ML = 5.5 + 0.013 \cdot TOW \cdot SO_2 \cdot O_3 \] [14]

To date, the assessments in the ExternE Project have not considered incremental ozone levels from fuel cycle emissions. These equations demonstrate that this may introduce additional uncertainty into our analysis.

11.4.8 Aluminium

Aluminium is the most corrosion resistant of the common building materials. In the atmosphere aluminium becomes covered with a thin, dense, oxide coating, which is highly protective down to a pH of 2.5. In clean outdoor atmospheres, aluminium will retain its appearance for years, even under tropical conditions.

The major pollution related damage mechanism of concern is pitting due to SO\(_2\) (Lipfert, 1987). However, no dose-response functions have been derived for this process. Surface corrosion is less of a problem. Although aluminium has a good corrosion resistance, increasing sulphur dioxide and nitrogen oxide concentrations will accelerate corrosion. There is a significant correlation between corrosion rate and SO\(_2\) concentration, but pollutant deposition does not seem to play such a dominant role as in the case of zinc. Other parameters, such as the time of wetness, may also have a significant influence on the corrosion rate of aluminium (Kucera et al., 1988; Dean and Anthony, 1988).

A few dose-response functions have been derived and there is general agreement that the kinetics are approximately linear. Meta-analysis (Lipfert, 1987) suggests a function of the form shown in equation [15]. The ICP study has also produced functions for aluminium samples after 4 years, shown in equations [16] and [17].

Lipfert - aluminium (annual loss):
\[ ML = 0.2 \cdot t^{0.99} \cdot (0.14 \cdot f_3 \cdot SO_2 + 0.093 \cdot Cl^- + 0.0045 \cdot H^+ - 0.0013 \cdot D)^{0.88} \] [15]

ICP - unsheltered aluminium (4 year):
\[ ML = 0.85 + 0.0028 \cdot TOW \cdot SO_2 \cdot O_3 \] [16]

ICP - sheltered aluminium (4 year):
\[ ML = -0.03 + 0.0053 \cdot TOW \cdot SO_2 \cdot O_3 + 74 \cdot Cl^- (p) \] [17]

11.4.9 Recommendations on the use of dose-response functions

The functions we have used in our analysis (both for the UK and German assessments) and the approach we recommend in future analyses are summarised below.
Stone
The UK assessments of stone erosion used all three studies, i.e. the Lipfert, Butlin and the Kucera limestone functions (equations [1], [2] and [3]). An average of the three functions was used for the best estimate, with upper and lower estimates taken directly from the range of values produced. No account of stage II and III damages were considered. It was assumed that all stone in the inventory was exposed, i.e. the proportion of sheltered material was not considered.

The analysis of the German plants differed slightly to this approach. Firstly, more detail was available for the German stone inventory, with the proportion of sandstone, limestone and other natural stone calculated. It was therefore possible to apply the individual functions from the Kucera functions for limestone and sandstone (equations [3] and [5]). This approach was also used for other Western and Southern European countries though with less certainty because of the extrapolation of the inventory. For the remaining ‘natural stone’ in the inventory, and for Eastern and Northern Europe, the Kucera sandstone functions was used (equation [5]). The analysis also considered the proportion of sheltered material, assuming a third of material was not exposed. For this material, the relevant ICP functions (equations [6] and [7]) were used. These calculate the mass increase on the stone surface though the critical thickness loss and repair costs used (see next section) mean a lower confidence is attached to these results.

The method and functions we recommend for future studies depend to a degree on obtaining improved inventories and a need for some basic research. Ideally, the stone inventory should be categorised by type into limestone, sandstone, etc. The ICP functions then provide the best functions for the analysis of European damages. Where available, studies, such as the UK NMEP functions should be fed into the analysis to produce a range of values for the assessment.

In cases where detailed inventory data is not available, a broader range of functions may be more applicable, including the use of the Lipfert function. In all cases, upper and lower limits should be defined using a range of functions. Given the large difference in damages between sheltered and unsheltered material, we recommend that the proportion of sheltered material be defined and the appropriate functions used. However, these estimates of sheltered damage costs should be treated with some care until more work is undertaken to categorise the exact proportion of sheltered material, and to determine the critical loss and repair costs for these impacts.

Mortar, rendering and concrete
To date, previous estimates for mortar damage have been identical to the stone implementation above, with the difference that the functions used have been three time larger to account for the calcareous portion. However, we now advise the use of the original functions for sandstone (equations [5] and [4]) to respectively calculate best estimate and the range of damages. The assessment of rendering should also follow this approach. At present, we do not recommend the use of any functions for the analysis of concrete damage.
Paint

The best estimate for UK paint damage was calculated using the function for carbonate paint. These carbonate based paints are rapidly being replaced by acrylic and epoxy resin based paint, particularly in the rest of Europe. Thus, for the analysis of European damage costs, we have applied the silicate paint function, though the carbonate function has also been considered to provide an upper limit. This silicate function was also used to calculate the lower estimate for the UK. A nominal factor of two from the best estimate was used to provide the other upper and lower values in the ranges. In the absence of better information, this seems a reasonable approach although as more information becomes available from European studies, the method should be updated. The material inventory for steel (all painted) and the relevant proportion of painted galvanised steel should be included in the paint inventories.

Galvanised steel and metals

In the UK studies of galvanised steel, we have used average values derived from equations [11] and [13]. The German studies only used the ICP function. Both functions have serious limitations for implementation; the transferability of the Lipfert function is questionable and the time function makes the implementation difficult, whilst the ICP function relies on a number of key parameters which are poorly characterised across Europe. The extension of the UK inventory to also consider general infrastructure is a step forward, though because of the lack of data from other countries, total European estimates should currently be treated with some caution. To date, no account has been taken of unsheltered material, though this should be considered for some sectors. Some care should be exercised in the quantity of material exposed. Galvanised steel that is painted before or during construction should be treated in the same way as other painted surfaces and this should be transferred to the paint inventory above.

The results of our analysis have shown the likely damage from pollution on aluminium to be negligible. However, in areas with very high pollutant concentrations, the analysis should be considered using average rates from equations [15] and [16].

11.5 Estimation of Impacts on Materials

11.5.1 Calculation of repair frequency

The information derived from the dose-response functions above must next be converted into a measure of damage to building material. This can prove very difficult. Most dose-response relationships are given in terms of weight or thickness loss as a function of time. To value impacts, these losses must be converted into repair or replacement frequencies. This is typically done using either engineering assessments or behavioural data on building maintenance practice, though there is a shortage of usable information for both approaches.
In this study, published information and expert assessment were used wherever possible. Where no information was available, estimates based on common experience were used. Whilst this approach will inevitably lead to some error and uncertainty, it is expected that the impacts derived will be well within an order of magnitude and are useful in determining the magnitude of the external cost. However, further work is required to reduce the uncertainty associated with the analysis. A summary of the critical thickness loss for maintenance and repair are shown in Table 11.6.

For natural stone and mortar, it is assumed that maintenance action will be required after 5 mm of surface is lost. However, the actual "critical loss" will depend on application. Some commentators have said that the use of erosion losses of this order are too low. However, it must be remembered that damage is typically not uniform across a surface, and that the timing of maintenance is likely to be determined by the condition of the worst affected areas. In view of this, the average loss above certainly seems to be justified. This critical loss value was also used for the repair frequency of render.

A similar critical level (3-5 mm) was used for the repair action for mass increase from crust build-up on sheltered materials in the German assessment, as no repair frequencies exist to allow accurate estimates of this impacts. In the absence of other data, this seems a sensible approach and provides a useful first estimate for sheltered material, though the confidence associated with this estimate is very low.

Because of the different inventory data, slightly different approaches for zinc and galvanised steel were used between project teams. For zinc components, which are used mainly for gutters and down pipes on dwelling houses, building components were assumed to be replaced when a thickness loss of 2 mm was reached. For galvanised steel, the thickness of the coating depends upon the galvanising process. Thicker coatings are generally obtained by the "dipping" techniques used for general galvanising compared to the continuous processes used for galvanising of wire and sheet. The latter are more numerous as construction materials. A coating thickness of 20 µm was used in the German implementation (NAPAP, 1990). It is assumed that galvanised steel is used where corrosion of the underlying mild steel is unacceptable. The repair frequency is therefore determined by the lifetime of the coating.

For the UK assessment, specific coating thicknesses were assigned for each individual sector analysed in the galvanised inventory (Short, 1994a), shown in Table 11.6 below. We have assumed galvanised steel repair would take place before total removal of the film. As an approximation, we have taken the value of 50% film erosion for the initiation of repair action. Importantly, we believe the most likely first repair action would be painting rather than replacement. For our upper estimate, we have assumed 50% of the material is repainted and 50% is replaced, though we assume complete removal of the zinc film is required for the latter action.

The dose-response functions for calcium carbonate containing paints were used to provide the change in paint erosion rate, based on observation. We have used a critical paint thickness of 20 µm (Haynie, 1986), defined to be the erosion required to produce coating failure. This
thickness has also been applied to the silicate paints used in the general European inventory. A nominal factor of two either side of this estimate gives a range for the analysis.

**Table 11.6 Critical Thickness Loss for Maintenance or Repair Measures.**

<table>
<thead>
<tr>
<th>Material</th>
<th>Critical thickness loss</th>
</tr>
</thead>
<tbody>
<tr>
<td>Natural stone</td>
<td>5 mm</td>
</tr>
<tr>
<td>Mortar 1</td>
<td>5 mm</td>
</tr>
<tr>
<td>Zinc</td>
<td>2 mm</td>
</tr>
<tr>
<td>Galvanised steel (general) 2</td>
<td>20 µm</td>
</tr>
<tr>
<td><strong>Galvanised steel by sector</strong></td>
<td></td>
</tr>
<tr>
<td>Dipped products</td>
<td></td>
</tr>
<tr>
<td>Building and construction</td>
<td>100 µm 3</td>
</tr>
<tr>
<td>Street furniture</td>
<td>100 µm 3</td>
</tr>
<tr>
<td>Power sector (electricity pylons)</td>
<td>200 µm 3</td>
</tr>
<tr>
<td>Transport infrastructure</td>
<td>100 µm 3</td>
</tr>
<tr>
<td>Agriculture</td>
<td>200 µm 3</td>
</tr>
<tr>
<td>Sheet and strip</td>
<td></td>
</tr>
<tr>
<td>Building and construction</td>
<td>25 µm 3</td>
</tr>
</tbody>
</table>

1 The German analysis used a range of 3-5 mm. For the UK implementation a 5 mm critical loss was used.

2 Used in the German implementation.

3 Individual sector inventory was used for the UK inventory, see Table 11.A5 in Appendix. Values shown represent actual coating thickness. For repair action, we assume repainting occurs when 50% of this total coating thickness is lost.

The predominant use of aluminium is in doors and windows. Even at the highest corrosion rates predicted by the dose-response function for UK conditions, loss of 100 µm would take 300 years, well in excess of the expected lifetime of the component. Pollution induced impacts are therefore negligible.

In the analysis of fossil fuel cycles, the estimates of repair frequency are made for the two cases; with and without the reference power station. The difference between these estimates represents the incremental damage associated with the reference power plant.

**11.5.2 Estimation of economic damage (repair costs)**

The valuation of impacts should ideally be made from the willingness to pay to avoid the incremental damage. No assessments of this type are available. Instead, repair/replacement costs of building components are used as a proxy estimate of economic damage.

For such estimates, an important issue concerns the variation in behaviour between individuals. Some people will allow damage to proceed well beyond the point at which action would ideally be taken, and suffer higher repair costs as secondary damage mechanisms take effect, perhaps even affecting the structural integrity of their property. Other people will take
Impacts on Materials

action earlier than what could be defined as the economic 'ideal' from the perspective of repair costs. This could be because they believe that a given action should be performed at a particular frequency, or, for example, simply because they want to paint their house a different colour. In such cases it is logical to assume that there is no economic effect of pollution damage. Quantification of the proportion of cases that may fall into either category is not currently possible. However, it is clear from the English House Condition Survey that a substantial number of dwellings (the most common type of building) require significant amounts of remedial work.

If owners and occupiers of buildings act in an economically rational way then the costs of additional repair and maintenance will be less than, or equal to, the damage costs. Under this approach additional building maintenance costs are a minimum valuation of the damage benefits.

It is necessary to make some assumptions about the timing of the costs. For a building stock with a homogeneous age distribution, the incidence of repair and replacement costs will be uniform over time, irrespective of the pollution level. The repair/replacement frequency is then an adequate basis for valuation with costs assumed to occur in the year of the emission. The reference environment building stock corresponds relatively well to the requirement of a homogeneous age distribution. There are some exceptions, where the age distribution, and consequently replacement time distribution, are more strongly concentrated in some periods. However, the error in neglecting this effect will be small compared to other uncertainties in the analysis.

Estimates of economic damage to natural stones, mortar, painted surfaces, zinc/galvanised steel, iron/steel and aluminium have been made and are shown in Table 11.7. The estimated repair costs for the UK analysis are taken from unit cost factors for each of the materials for which assessment was performed. These figures are based on data from ECOTEC (1986) and Lipfert (1987). The repair costs used in the analysis of impacts from German plants have been obtained by inquiries with German manufacturers. Finally, damage costs given in a study for Stockholm, Prague and Sarpsborg (Kucera et al, 1993b) are included.

Table 11.7 Comparison of Damage Costs for Materials Under Analysis.

<table>
<thead>
<tr>
<th>Repair or maintenance category</th>
<th>UK 1 (ECU/m²)</th>
<th>German 2 (ECU/m²)</th>
<th>Kucera 3 (ECU/m²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Repainting</td>
<td>11.3</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Replacing zinc</td>
<td>/</td>
<td>22</td>
<td>/</td>
</tr>
<tr>
<td>Replacing galvanised steel</td>
<td>45.1</td>
<td>25</td>
<td>47.7</td>
</tr>
<tr>
<td>Refacing natural stone</td>
<td>248</td>
<td>190 - 300</td>
<td>/</td>
</tr>
<tr>
<td>Repointing</td>
<td>22.8</td>
<td>27</td>
<td>50.4</td>
</tr>
</tbody>
</table>

1 Converted from £1990 to 1990 ECU using a conversion rate of £1 = 1.4 ECU.
2 Converted from 1990 DM to 1990 ECU using a conversion rate of 1DM = 0.5 ECU.
3 Converted using a conversion rate of 1 Schwedische Krone = 0.265 DM and 1 Sw. Krone = 0.133 ECU.
We propose to use country specific repair costs where available. However, the lack of information for most of Europe needs to be redressed with further research into repair costs for different regions.

It should be noted that we have applied identical repair costs for all types of repainting, whether on wood surfaces, steel, galvanised steel, etc. This is likely to underestimate impacts, as some paints such as the zinc rich coatings applied to galvanised steel will be more expensive.

To obtain a damage estimate for the reference environment, the frequency of replacement and the total area of each material in each grid cell were multiplied by these unit cost factors. The same was done to calculate the costs with the incremental pollution due to the power plant. The change in cost for each material was calculated by subtracting one from the other. Changes were then summed to give the total economic damage.

11.5.3 Estimation of soiling costs

Soiling of buildings results primarily from the deposition of particulates on external surfaces. Three major categories of potential damage cost may be identified; damage to the building fabric, cleaning costs and amenity costs. In addition, there may be effects on building asset values, as a capitalised value of these damages.

There are a number of sources of particulate emissions from fossil fuel cycles although the largest emission will arise from the power station stack. We have not considered the effect of secondary particulates (acid aerosols) on soiling, though whether this will underestimate damages is not known. The role of particulates as catalysts for the conversion of SO₂ and NOₓ to their respective acids (enhancing stone erosion) is also not considered. We therefore limit this assessment to the deposition of particulates on building surfaces, which results in a darkening of the material; for stone this can also include incorporation into gypsum crusts on sheltered surfaces.

Cleaning costs and amenity costs need to be considered together. Data on the former is, of course, easier to identify. In an ideal market, the marginal cleaning costs should be equal to the marginal amenity benefits to the building owner or occupier. However, markets are not perfect and amenity benefits to the public as a whole lie outside this equation. It is therefore clear that cleaning costs will be lower than total damage costs resulting from the soiling of buildings. In the absence of willingness to pay data, cleaning cost are used here as an indicator of minimum damage costs.

The range of primary particulates should really be assessed over a regional scale, though the larger particulates will have a shorter range than other atmospheric pollutants (see Figure 3.1). Dose-response functions for particulate soiling (in terms of reduced surface reflectance) of a variety of materials were given by Beloin and Haynie (1975). More recent functions have been proposed for soiling (Hamilton and Mansfield, 1992) which predict the % change in reflectance for exposed painted wood and sheltered painted wood. However, the estimate of
associated damages given in this section is too low to warrant detailed analysis using dose-response functions.

Instead, we have taken a simple approach to try and derive soiling costs. For example, in the analysis of UK plants, we assume that the total impact of building soiling will be experienced in the UK. The total UK building cleaning market is estimated to be £80 million annually (Newby et al., 1991). Most of this is in urban areas and it is assumed that it is entirely due to anthropogenic emissions. Moreover, it can reasonably be assumed that cleaning costs are a linear function of pollution levels, and therefore that the marginal cost of cleaning is equal to the average cost.

Different types of particulate emission have different soiling characteristics (Newby et al., 1991). The appropriate measure of pollution output is therefore black smoke, which includes this soiling weighting factor, rather than particulates, which does not. UK emissions of black smoke in 1990 were 453,000 tonnes (DOE, 1991). The implied average marginal cost to building cleaning is therefore £180/tonne. This value is simply applied to the plant output. The method assumes that emission location is not important; in practice, emissions from a plant outside an urban area will have a lower probability of falling on a building. However, given the low magnitude of the impact, this method provides an approximate value.

Results from the French implementation (European Commission, 1995) have shown that for particulate soiling, the total cost is the sum of repair cost and the amenity loss. The results show that, for a typical situation where the damage is repaired by cleaning, the amenity loss is equal to the cleaning cost (for zero discount rate); thus the total damage costs is twice the cleaning cost. Data from the same study shows cleaning costs for other European countries may be considerably higher than the UK values. Further estimates of cleaning costs need to be assimilated so that these impacts can be assessed in a more detailed way.

11.6 Conclusions

This chapter describes the methodology we have used to analyse the impacts of acidic deposition on materials. The approach assesses damages to the common building materials used across Europe, deriving economic costs for the material loss resulting from atmospheric pollution levels.

The method uses inventories of the stock of material at risk, built-up from building surveys in several European cities and extrapolated to the whole of Europe using population data. The pollution data from atmospheric models is then used with the most up to date dose-response functions to calculate the surface erosion on exposed material. These losses are then assessed using data on repair and replacement frequencies, and the costs of such repair actions, to produce economic damages.
Many uncertainties remain in the analysis. In particular, the total damage cost derived is sensitive to some parts of the analysis which are rather uncertain and require further examination. The following are identified as research priorities:

- Improvement of inventories, in particular; the inclusion of country specific data for all parts of Europe; disaggregation of the inventory for paint to describe the type of paint in use; disaggregation of the inventory for galvanised steel to reflect different uses; disaggregation of calcareous stone into sandstone, limestone, etc. In addition, alternatives to the use of population data for extrapolation of building inventories should be investigated.
- Further development of dose-response functions, particularly for paints, mortar, cement render, and of later, more severe damage mechanisms on stone;
- Assessment of exposure dynamics of surfaces of differing aspect (horizontal, sloping or vertical), and identification of the extent to which different materials can be considered to be sheltered;
- Definition of service lifetimes for stone, concrete and galvanised steel;
- Integration of better information on repair techniques;
- Data on cleaning costs across Europe;
- Improvement of awareness of human behaviour with respect to buildings maintenance;
- The extension of the methodology for O₃ effects, including development of dose-response functions and models atmospheric transport and chemistry.

Although this list of uncertainties is extensive, it would be wrong to conclude that our knowledge of air pollution effects on buildings is poor, certainly in comparison to our knowledge of effects on many other receptors. Indeed, we feel that the converse is true; it is because we know a great deal about damage to materials that we can specify the uncertainties in so much detail.

Some of these uncertainties will lead to an underestimation of impacts, and some to an overestimation. The factors affecting galvanised steel are of most concern given that damage to it comprises such a high proportion of total materials damage. However, a number of potentially important areas were excluded from the analysis because no data were available. In general, inclusion of most of these effects would lead to greater estimates of impacts. They include:

- Effects on historic buildings and monuments with "non-utilitarian" benefits;
- Damage to utilitarian structures that were not included in the inventory;
- Damage to paint work through mechanisms other than acid erosion;
- Damage to reinforcing steel in concrete;
- Synergies between different pollutants;
- Impacts of emissions from within Europe on buildings outside Europe;
- Impacts from ozone.
11.7 References


Statistisches Bundesamt (StaBA) (Eds.): Länderberichte. Wiesbaden, Several Years.


## Appendix 11.1 Building Identikits Used for the Implementations

### Table 11.A1 Building Identikit 2 Used to Describe the Use of Different Types of Materials Used in the Construction of Schools.

<table>
<thead>
<tr>
<th>Building identikit 2</th>
<th>Schools</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flat Roof (m²)</td>
<td>1861.63</td>
</tr>
<tr>
<td>Pitched Roof (m²)</td>
<td>0</td>
</tr>
<tr>
<td>Wall Area (m²)</td>
<td>723.66</td>
</tr>
<tr>
<td>Windows and Doors (m²)</td>
<td>377.1</td>
</tr>
<tr>
<td>Drainage (m²)</td>
<td>0</td>
</tr>
<tr>
<td>Total External Envelope (m²):</td>
<td>2962.39</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Predominant materials</th>
<th>Model (%)</th>
<th>Quantity (m²/Building)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flat Roofs:</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Asphalt on Screed</td>
<td>1</td>
<td>1861.63</td>
</tr>
<tr>
<td>Walls:</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Concrete Frame Elements</td>
<td>0.01</td>
<td>7.16</td>
</tr>
<tr>
<td>Bare Brickwork</td>
<td>0.83</td>
<td>603.37</td>
</tr>
<tr>
<td>Timber Cladding</td>
<td>0.16</td>
<td>113.13</td>
</tr>
<tr>
<td>Windows and Doors:</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Painted Joinery</td>
<td>0.5</td>
<td>188.55</td>
</tr>
<tr>
<td>Alum/Metal/UPVC</td>
<td>0.5</td>
<td>188.55</td>
</tr>
<tr>
<td><strong>Total:</strong></td>
<td></td>
<td><strong>2962.39</strong></td>
</tr>
</tbody>
</table>
Table 11.A2  Building Identikit 3 Used to Describe the Use of Different Types of Materials Used in the Construction of Shops.

<table>
<thead>
<tr>
<th>Building identikit 3</th>
<th>Shops</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flat Roof (m²)</td>
<td>59.73</td>
</tr>
<tr>
<td>Pitched Roof (m²)</td>
<td>21.23</td>
</tr>
<tr>
<td>Walls (m³)</td>
<td>53.03</td>
</tr>
<tr>
<td>Windows and Doors (m²)</td>
<td>23.76</td>
</tr>
<tr>
<td>Drainage (m²)</td>
<td>5.28</td>
</tr>
<tr>
<td><strong>Total External Envelope (m²):</strong></td>
<td><strong>163.03</strong></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Predominant materials</th>
<th>Model (%)</th>
<th>Quantity (m²/Building)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Flat Roofs:</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Asphalt on Screed</td>
<td>1</td>
<td>59.73</td>
</tr>
<tr>
<td><strong>Pitched Roofs:</strong></td>
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<tr>
<td>Clay Tiles</td>
<td>0.3</td>
<td>6.37</td>
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<tr>
<td>Slate Tiles</td>
<td>0.6</td>
<td>12.74</td>
</tr>
<tr>
<td>Concrete Tiles</td>
<td>0.1</td>
<td>2.12</td>
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<tr>
<td><strong>Walls:</strong></td>
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<td></td>
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<tr>
<td>Concrete Frame Elements</td>
<td>0.02</td>
<td>1.16</td>
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<tr>
<td>Bare Brickwork</td>
<td>0.46</td>
<td>24.52</td>
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<tr>
<td>Painted Brickwork</td>
<td>0.04</td>
<td>2.23</td>
</tr>
<tr>
<td>Stone</td>
<td>0.1</td>
<td>5.43</td>
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<td>Render</td>
<td>0.23</td>
<td>12.3</td>
</tr>
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<tr>
<td>GRP/Galvanised Steel Cladding</td>
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<td>Timber Cladding</td>
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<td>Plastic Board Cladding</td>
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<tr>
<td><strong>Windows and Doors:</strong></td>
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<td></td>
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<tr>
<td>Painted Joinery</td>
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<td>5.93</td>
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<tr>
<td>Alum/Metal/UPVC</td>
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<td>17.83</td>
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<tr>
<td><strong>Drainage:</strong></td>
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<tr>
<td>Plastic</td>
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<tr>
<td>Cast Iron</td>
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</tr>
<tr>
<td><strong>Total:</strong></td>
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<td>163.03</td>
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Table 11.A3  Building Identikit 4 Used to Describe the Use of Different Types of Materials Used in the Construction of Industrial Buildings.

<table>
<thead>
<tr>
<th>Building identikit 4</th>
<th>Industrial</th>
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</thead>
<tbody>
<tr>
<td>Flat Roof (m²):</td>
<td>6062.55</td>
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<tr>
<td>Pitched Roof (m²):</td>
<td>0</td>
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<td>Walls (m²):</td>
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<td>Windows and Doors (m²):</td>
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<tr>
<td>Drainage (m²):</td>
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<tr>
<td>Total External Envelope (m²):</td>
<td>7786.55</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Predominant materials</th>
<th>Model (%)</th>
<th>Quantity (m²/Building)</th>
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<tr>
<td><strong>Flat Roofs:</strong></td>
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<td></td>
</tr>
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<td>Asphalt on Screed</td>
<td>0.22</td>
<td>1345</td>
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<td>Asbestos Cement Tiles</td>
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<td>1506.1</td>
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<td>Asbestos Cement Sheet</td>
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<td>Bare Brickwork</td>
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<td>Metal Cladding</td>
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<td>GRP/Galvanised Steel</td>
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<td>596.847</td>
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<tr>
<td><strong>Windows and Doors:</strong></td>
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<td></td>
</tr>
<tr>
<td>Painted Joinery</td>
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<td>2.6</td>
</tr>
<tr>
<td>Alum/Metal/UPVC</td>
<td>0.9</td>
<td>23.4</td>
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<td><strong>Total:</strong></td>
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Table 11.A4  Building Identikit 5 Used to Describe the Use of Different Types of Materials Used in the Construction of Offices.

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<th>Offices</th>
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</thead>
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<td>Flat Roof (m²):</td>
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<td>Pitched Roof (m²):</td>
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</tr>
<tr>
<td>Walls (m²):</td>
<td>211</td>
</tr>
<tr>
<td>Windows and Doors (m²):</td>
<td>158</td>
</tr>
<tr>
<td>Drainage (m²):</td>
<td>0</td>
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<tr>
<td>Total External Envelope (m²):</td>
<td>648.6</td>
</tr>
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</table>

<table>
<thead>
<tr>
<th>Predominant materials</th>
<th>Model (%)</th>
<th>Quantity (m²/Building)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Pitched Roofs:</strong></td>
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<td></td>
</tr>
<tr>
<td>Asphalt on Screed</td>
<td>0.575</td>
<td>160.8</td>
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<tr>
<td>Clay Tiles</td>
<td>0.035</td>
<td>9.45</td>
</tr>
<tr>
<td>Slate Tiles</td>
<td>0.39</td>
<td>109.35</td>
</tr>
<tr>
<td><strong>Wall Area:</strong></td>
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<td></td>
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<tr>
<td>Concrete Panels</td>
<td>0.48</td>
<td>100.7525</td>
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<td>Bare Brickwork</td>
<td>0.2</td>
<td>42.2</td>
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<tr>
<td>Stone</td>
<td>0.11</td>
<td>22.6825</td>
</tr>
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<td>Curtain Walling</td>
<td>0.21</td>
<td>45.365</td>
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<td><strong>Windows and Doors:</strong></td>
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<td>Stone</td>
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</tr>
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<td>Painted Joinery</td>
<td>0.485</td>
<td>76.5</td>
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<td>79.6</td>
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<tr>
<td><strong>Total:</strong></td>
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<td>648.6</td>
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</table>
Table 11.A5 Galvanised Steel Stock at Risk for the UK.

<table>
<thead>
<tr>
<th>Sector</th>
<th>Zn used (tonnes)</th>
<th>Thickness (µm)</th>
<th>% Area exposed</th>
<th>Range exposed</th>
<th>Surface area (10^6m²/yr)</th>
<th>Stock at risk (10^6m²/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>General galv.</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Construction</td>
<td>13,600</td>
<td>100 µm</td>
<td>20</td>
<td>10 - 50%</td>
<td>3.81</td>
<td>57.14</td>
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<tr>
<td>Street furniture</td>
<td>10,000</td>
<td>100 µm</td>
<td>50</td>
<td>30 - 70%</td>
<td>7.00</td>
<td>105.04</td>
</tr>
<tr>
<td>Power</td>
<td>5,200</td>
<td>200 µm</td>
<td>100</td>
<td>50 - 100%</td>
<td>3.64</td>
<td>54.62</td>
</tr>
<tr>
<td>Agriculture</td>
<td>3,200</td>
<td>100 µm</td>
<td>75</td>
<td>50 - 100%</td>
<td>3.36</td>
<td>50.42</td>
</tr>
<tr>
<td>Transport</td>
<td>1,200</td>
<td>200 µm</td>
<td>75</td>
<td>50 - 100%</td>
<td>0.63</td>
<td>9.45</td>
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<tr>
<td>Fasteners 1</td>
<td>1,600</td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Others 1</td>
<td>5,200</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sheet and strip</td>
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</tr>
<tr>
<td>Construction 2</td>
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<td>48.32</td>
<td>253.93</td>
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<td>Automobiles 1</td>
<td>15,500</td>
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<td>Miscellaneous 1</td>
<td>12,000</td>
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<td>Appliances 1</td>
<td>2,500</td>
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<td>Wire 1</td>
<td>14,000</td>
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<tr>
<td>Tube 1</td>
<td>3,000</td>
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<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

1 Not considered in our analysis.
2 The values from the ECOTEC study were used, as they are thought to be more accurate.
3 From estimated quantity painted.
4 The 1990 data was multiplied up to give a 20 year stock at risk. A factor of 15 was used due to the increase in steel production since the 1970s. A density of 7.14 tonnes/m³ is used to give the best estimate of surface area.
5 The 20 year stock is assumed by multiplying 1990 figures by 15.
### Table 11.A6 Materials at Risk in Dortmund

<table>
<thead>
<tr>
<th>Settlement area</th>
<th>ST 1</th>
<th>ST 2</th>
<th>ST 3</th>
<th>ST 4</th>
<th>ST 5</th>
<th>ST 6</th>
<th>ST 7</th>
<th>ST 8</th>
<th>ST 9</th>
</tr>
</thead>
<tbody>
<tr>
<td>in % of town area</td>
<td>2.97</td>
<td>16.82</td>
<td>4.4</td>
<td>4.47</td>
<td>0.7</td>
<td>2.97</td>
<td>0.41</td>
<td>0</td>
<td>5.69</td>
</tr>
<tr>
<td>absolute (1000 m²)</td>
<td>6880</td>
<td>39030</td>
<td>10210</td>
<td>10370</td>
<td>1620</td>
<td>6880</td>
<td>940</td>
<td>0</td>
<td>13200</td>
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<tr>
<td>Number of buildings</td>
<td>3509</td>
<td>39811</td>
<td>15315</td>
<td>7778</td>
<td>851</td>
<td>8256</td>
<td>1410</td>
<td>/</td>
<td>3960</td>
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</table>

**Envelope elements (1000 m²)**

<table>
<thead>
<tr>
<th></th>
<th>Wall</th>
<th>Window</th>
<th>Roof</th>
</tr>
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<tbody>
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<td></td>
<td>514.6</td>
<td>93.6</td>
<td>514.6</td>
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<tr>
<td></td>
<td>5905.2</td>
<td>663.5</td>
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<td>1051.6</td>
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<tr>
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<td>2737.7</td>
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<td>1410.3</td>
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<td>243.0</td>
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<td>284.8</td>
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<tr>
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<td>1716.0</td>
<td>924.0</td>
<td>3960.0</td>
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</table>

**Surfaces (1000 m²)**

<table>
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<tr>
<th></th>
<th>Galvanised steel</th>
<th>Mild steel</th>
<th>Aluminium</th>
<th>Sandstone</th>
<th>Limestone</th>
<th>Concrete</th>
<th>Oil-based house paint</th>
<th>Acrylic paint</th>
<th>Latex-based paint</th>
<th>Rendering</th>
<th>Br. Facing</th>
<th>Asb. cement</th>
<th>Natural stone¹</th>
<th>Plastics</th>
<th>C.R.T.²</th>
<th>C.I.R.T.³</th>
<th>F.I.R.M.⁴</th>
<th>Zinc</th>
<th>Others</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>17.5</td>
<td>/</td>
<td>11.7</td>
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<td>35.1</td>
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<td>228.1</td>
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<td>17.5</td>
<td>100.5</td>
<td>421.1</td>
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<td>64.3</td>
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<td>66.4</td>
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<td>30.6</td>
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<td>/</td>
<td>/</td>
<td>/</td>
<td>/</td>
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<td>172.0</td>
<td>/</td>
<td>970.1</td>
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<td>/</td>
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</tr>
</tbody>
</table>

1: Natural stone other than lime- and sandstone.
2: Cement roofing tile.
3: Clay roofing tile.
4: Flat roof comprising bituminous and tar-based substances.
5: Zinc and its alloys.

Source: Hoos et al. (1986).
Table 11.A7 Materials at Risk in Köln

<table>
<thead>
<tr>
<th>Settlement area</th>
<th>ST 1</th>
<th>ST 2</th>
<th>ST 3</th>
<th>ST 4</th>
<th>ST 5</th>
<th>ST 6</th>
<th>ST 7</th>
<th>ST 8</th>
<th>ST 9</th>
</tr>
</thead>
<tbody>
<tr>
<td>in % of town area</td>
<td>4.06</td>
<td>8.01</td>
<td>2.17</td>
<td>4.53</td>
<td>0.54</td>
<td>6.26</td>
<td>0.85</td>
<td>0.15</td>
<td>6.23</td>
</tr>
<tr>
<td>(sum 33.66 %)</td>
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<td></td>
<td></td>
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<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>absolute (1000 m²)</td>
<td>12600</td>
<td>24820</td>
<td>6730</td>
<td>14040</td>
<td>1660</td>
<td>19420</td>
<td>2620</td>
<td>450</td>
<td>19310</td>
</tr>
<tr>
<td>Number of buildings</td>
<td>6426</td>
<td>25316</td>
<td>10095</td>
<td>10530</td>
<td>872</td>
<td>23304</td>
<td>3930</td>
<td>1530</td>
<td>5793</td>
</tr>
</tbody>
</table>

| Envelope elements (1000 m²) | | | | | | | | | |
| Wall                       | 942.5| 3755.3| 693.2| 3706.6| 373.5| 6233.8| 793.9| 265.5| 2510.3|
| Window                     | 171.4| 421.9| 154.8| 926.6| 249.0| 1553.6| 264.6| 88.7 | 1351.7|
| Roof                       | 942.5| 2890.3| 888.4| 1909.4| 102.9| 3495.6| 681.2| 236.3| 5793.0|

| Surfaces (1000 m²)        | | | | | | | | | |
| Galvanised steel          | 32.1| 84.4| 26.9| 56.2| / | 97.1| 13.1| 7.2 | 77.2 |
| Mild steel                | / | / | / | / | / | / | / | / | / |
| Aluminium                 | 21.4| 42.2| 20.2| 126.4| 43.2| 38.8| 188.6| / | 675.9 |
| Sandstone                 | / | / | / | / | 58.3| 49.8| 40.1| / | / |
| Limestone                 | / | / | / | / | / | / | / | / | 94.3 |
| Concrete                  | 21.4| 84.4| 13.5| 407.2| 142.8| / | 47.2| / | 405.5 |
| Oil-based house paint     | 64.3| 168.8| 208.6| 224.6| 102.9| 485.5| 41.9| 54.0| 57.9 |
| Acrylic paint             | 32.1| 126.6| 20.2| 70.2| 10.0| 194.2| 15.7| 10.8| 77.2 |
| Latex-based paint         | 417.7| 1645.6| 302.9| 884.5| 116.2| 2563.4| 277.7| 143.6| 926.9 |
| Rendering                 | 192.8| 780.6| 141.3| 2035.8| / | 2738.2| 79.6| / | 173.8 |
| Br. Facing                | 149.9| 590.7| 107.7| / | / | / | / | / | / |
| Asb. cement               | / | / | / | / | / | / | / | / | 1255.1|
| Natural stone¹            | / | / | / | / | / | / | 62.9| / | / |
| Plastics                  | 32.1| 63.3| 107.7| 196.6| / | 466.1| 21.0| / | 77.2 |
| C.R.T.²                   | 771.1| 2362.9| 726.8| 1530.4| / | 1223.5| 238.4| / | 695.2 |
| C.I.R.T.³                 | 42.8| 147.7| 47.1| / | / | 2272.1| / | / | 231.7 |
| Fl.R.M.⁴                  | 117.8| 379.7| 114.4| 379.1| 104.6| / | 442.8| / | 3939.2|
| Zinc                      | 139.2| 443.0| 101.0| 337.0| 8.3| 699.1| 81.2| 47.3| 347.6 |
| Others                    | 139.2| 548.5| 114.4| 294.8| 101.3| 893.3| 107.4| 18.9| 482.8 |

1: Natural stone other than lime- and sandstone.
2: Cement roofing tile.
3: Clay roofing tile.
4: Flat roof comprising bituminous and tar-based substances.
5: Zinc and its alloys.

### Table 11.A8  Material Inventories for Different European Regions (in Million m²)

<table>
<thead>
<tr>
<th>Material Type</th>
<th>Germany</th>
<th>Western/Southern Europe</th>
<th>UK</th>
<th>Eastern Europe</th>
<th>Northern Europe</th>
</tr>
</thead>
<tbody>
<tr>
<td>Natural stone (Sandstone)</td>
<td>33</td>
<td>51</td>
<td>9</td>
<td>/</td>
<td>/</td>
</tr>
<tr>
<td></td>
<td>(15)</td>
<td>(24)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Natural stone (Limestone)</td>
<td>(10)</td>
<td>(16)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>(7)</td>
<td>(11)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Zinc</td>
<td>179</td>
<td>495</td>
<td>/</td>
<td>984</td>
<td>179</td>
</tr>
<tr>
<td>Galvanised steel</td>
<td>58</td>
<td>91</td>
<td>1001</td>
<td>1476</td>
<td>58</td>
</tr>
<tr>
<td>Rendering</td>
<td>962</td>
<td>1501</td>
<td>/</td>
<td>2830</td>
<td>962</td>
</tr>
<tr>
<td>Mortar</td>
<td>/</td>
<td>/</td>
<td>1979</td>
<td>/</td>
<td>/</td>
</tr>
</tbody>
</table>
12. GLOBAL WARMING

12.1 Introduction

There is reason to believe that the effects of global climate change are the most serious of all the priority impacts of the fossil fuel cycles. However, analysis of the effects of global warming is complicated by a variety of factors including the long time scales involved, the global nature of impacts and uncertainty concerning the extent of global temperature increase and its effect on regional climate. To estimate the damages from global warming, two approaches are being followed in the ExternE Project; a simple literature review based approach and an impact pathway based approach.

The literature review approach assesses the contribution of reference power plant emissions to overall greenhouse gas concentrations. However, in contrast with other impacts analysed within the ExternE Project, there is no quantitative assessment of these impacts. Instead, external cost estimates are simply based on the existing literature of damages arising from climate change (e.g. Cline, 1992; Fankhauser, 1993; Hohmeyer and Gärtner, 1992; Tol, 1995). In general, these studies assess damages due to a level of global warming resulting from an equivalent doubling of pre-industrial carbon dioxide concentrations (2xCO₂); they therefore have to be scaled to determine the impacts of individual fuel cycles.

Although this methodology is less sophisticated than an impact pathway approach, it has enabled some estimates to be made; these estimates are included in the individual fuel cycle reports (European Commission, 1995a-c). Because the approach differs from the methodology used for other pollution impacts, and because they are based on impact analyses which have not been scrutinised in our project, they are considered to be only illustrative and are not recommended outputs of the ExTERN Project. Furthermore, the estimates are highly dependent on the exact literature source used.

The alternative method, using an impact pathway approach, is currently under development within the project. It seeks to follow, as closely as is possible, the general methodology of ExternE. Therefore, it is the marginal impacts of the fuel cycle which are assessed. If this is to be done in a coherent manner, it requires the definition of an underlying scenario of climate change. Although climate change is a truly global effect, the impacts will not be experienced equally over the whole world. General Circulation Models (GCMs) of the atmosphere can define changes to temperature and precipitation in a spatially disaggregated framework. Despite the uncertainties in regional changes, this is preferable to an assumption of no geographical variation.

The impacts (for the key sectors) are assessed using simple, but robust models of the physical, social and economic impacts of changes in temperature, rainfall and sea level. Damage assessment is then undertaken in the usual way.
These two approaches are described in greater detail in the following sections.

12.2 Methodology Based on Literature Review

The initial stage of this approach follows other impact pathways within the study. The incremental emissions of greenhouse gases associated with each fuel cycle are estimated from a review of the technologies. As the impacts are truly global, there is no need to distinguish between different geographical sources. Following this, the effects of these emissions on atmospheric radiative forcing, and resulting changes in mean global atmospheric temperature are modelled.

Although this stage is relatively well understood, there remain considerable uncertainties, particularly for:
- The feedback mechanisms within the climate system (which determine the relationship between radiative forcing and global equilibrium temperature);
- The effectiveness of the oceans in slowing changes in global temperature.

For these reasons, it is not necessary to use a detailed "state of the art" climate model for this project. Instead "user friendly" and flexible tools have been used.

The final step is to quantify climate change damages, using estimates of total damages presented in current literature, and to scale these damages according to the incremental warming calculated for the particular fuel cycle.

12.2.1 Greenhouse gas emissions

The first issue which must be resolved concerns which emissions should be included in the assessment of global warming. Carbon dioxide, methane and nitrous oxide are all known to contribute directly to the greenhouse effect (Houghton et al., 1990), and therefore are included. Other potential emissions are more complicated to define.

NO\textsubscript{x} may also contribute to global warming due to its role in the formation of tropospheric ozone, which is also a greenhouse gas. Data from the IPCC scientific assessment (Houghton et al., 1990), indicates that NO\textsubscript{x} could be significant in the years immediately following the emission. However, more recent work (Johnson and Henshaw, 1991) has indicated that the IPCC values are in error by a large factor. In addition, NO\textsubscript{x} emissions are likely to result in higher tropospheric concentrations of the OH radical which in turn will reduce the atmospheric lifetime of methane, producing a global cooling effect (Houghton et al., 1990). No definitive quantification of the magnitude of this effect is available, but some preliminary assessments (Eyre and Michaelis, 1991) indicate that it could outweigh the effects of ozone production. In any event, at the emissions' levels typical of fossil fuel cycles, NO\textsubscript{x} will contribute at least an order of magnitude less than CO\textsubscript{2} to global warming. Because of this, and the large uncertainties involved, the effects of NO\textsubscript{x} have not been included in this initial analysis.
The effects of SO$_2$ are also uncertain. Whilst SO$_2$ is not a greenhouse gas, its role in the formation of sulphate aerosol has recently received much attention. Aerosols reflect incoming solar radiation and therefore tend to cool the troposphere. The contribution of this effect to observed changes in radiative forcing has yet to be calculated with any precision, but some estimates (Houghton et al, 1992) indicate that the historical effect may well be a significant fraction of the anthropogenic effect. The non-uniform geographical distribution of sulphate aerosols complicates assessment of their effects. However, it is clear that the short atmospheric lifetime of the aerosols will confine effects to the short term, and that the much reduced levels of SO$_2$ in emissions from modern power stations will reduce future effects of this type. Because of these factors and the high levels of uncertainty, the global cooling effects of SO$_2$ are also not included in this initial analysis.

In summary, only the effects of emissions of carbon dioxide, methane and nitrous oxide are considered at this stage. The other emissions which may have a role in climate change, notably NO$_x$, SO$_2$ and VOCs have not been assessed in the literature based approach. They are, however, included in the latter impact pathway approach. Nevertheless, there are strong grounds to believe that the overall warming effect calculated will be in error by less than 10% due to omission of the effects of NO$_x$ and SO$_2$.

12.2.2 Atmospheric modelling

Most modelling exercises concerned with global warming seek to determine the overall climate change from different scenarios of major greenhouse gas emissions (Houghton et al, 1990). For the purposes of this project, the requirement is to assess the incremental effects of rather small changes in global emissions. For this reason it is not possible to use the existing published work to directly calculate the required results.

Instead, two alternative modelling approaches have been used. The first is a simple spreadsheet approach, developed within the ExternE Project, which calculates the decay of incremental greenhouse gas concentrations and the resulting warming effects over 300 years. This involves a number of approximations, but allows a flexible and transparent approach. The other is an existing climate modelling package; the STUGE modal (Wigley et al, 1991). Although this allows the use of improved scientific assessment, the package is not being used for the objective it was originally designed for. One result of this is the model only produces output over 100 years. For this reason, the final warming profile adopted is a combination of the two approaches, using the STUGE output for the first 100 years and the spreadsheet results, normalised to STUGE at t=100 years, for the following 200 years. The two approaches are described in the following sections.

The spreadsheet approach

The atmospheric concentrations of carbon dioxide, methane and nitrous oxide due to a pulse of emissions in a given year can be calculated provided that the decay rate in the atmosphere is known. For both methane and nitrous oxide, there is a single dominant chemical removal mechanism, which in an otherwise unperturbed atmosphere produces an exponential decay.
The characteristic lifetimes are 10 years for methane and 150 years for nitrous oxide (Houghton et al, 1990).

For carbon dioxide, the principal mechanism for removal from the atmosphere is by solution into the oceans. The oceans are stratified, and hence the controlling processes for CO$_2$ removal are the physical, chemical and biological mechanisms by which carbon moves within them. The IPCC assessment (Houghton et al, 1990) used a complex multi-box model for this system. However, it is possible to reproduce their results for the decay of a pulse of carbon dioxide (Houghton et al, 1990, Figure 1.2) with high accuracy using a simpler model (Derwent, 1990). This has the form:

$$f = a_1 + a_2 \exp(-t/T_2) + a_3 \exp(-t/T_3) + a_4 \exp(-t/T_4),$$

where $f$ is the fraction of carbon dioxide in the atmosphere after a time $t$ years. For this study, the long term retention fraction, $a_1$, is derived from an oceanic model of carbon dioxide uptake (Siegenthaler, 1983). The remaining parameters are derived by empirically fitting the function to the decay curve given in the IPCC Report (Houghton et al, 1990). The values of the constants are as follows:

- $a_1 = 0.14$
- $a_2 = 0.3$, $T_2 = 400$
- $a_3 = 0.3$, $T_3 = 50$
- $a_4 = 0.26$, $T_4 = 5$.

The direct contributions of methane and nitrous oxide to radiative forcing, relative to carbon dioxide, are 58 and 206 (on a mass basis) respectively (Houghton et al, 1990). Incorporating these factors allows the emissions retained in the atmosphere in any given year to be given in terms of carbon dioxide equivalent. It should be noted at this point that this conversion is only strictly valid at current concentration levels. As concentration levels rise, the marginal effect of carbon dioxide changes logarithmically (Houghton et al, 1990), whilst those of methane and nitrous oxide have a quadratic form. However, in the context of this exercise (where methane and nitrous oxide make relatively small contributions), the error introduced by using constant forcing factors is very small.

In order to convert carbon dioxide mass into concentration, a conversion factor of $1.28 \times 10^{-16}$ ppmv/g is used (Houghton et al, 1990). Radiative forcing is then calculated using a linear approximation to the logarithmic representation given by the IPCC (Houghton et al, 1990) of the form:

$$F = 6.3 \times \frac{dC}{C_0}$$

where $F$ is the change in radiative forcing (in W/m$^2$), $dC$ is the change in atmospheric concentration of CO$_2$ and $C_0$ is the initial concentration.

Provided that climate feedback factors are neglected, the equilibrium temperature rise due to a change in radiative forcing can be derived directly from the change in temperature required to maintain the radiative thermal equilibrium of the Earth (Houghton et al, 1990). The relationship is of the form:

$$T = 0.3 \times F$$
where $T$ is the temperature rise in Kelvin. This would give a global equilibrium temperature rise of 1.2 K. However, the IPCC “best guess” value (Houghton et al, 1990), including feedbacks is 2.5 K, within a range of 1.5-4.5 K. The spreadsheet therefore incorporates a factor of 2.5/1.2 to ensure that the marginal temperature rises are consistent with the IPCC “best guess”.

These spreadsheet calculations provide a transparent and flexible method for calculating the equilibrium temperature rise resulting from fuel cycle emissions as a function of time. It is clear from the discussion above that some important approximations have been made in order to use this approach. In particular, the carbon cycle model is very crude, making no allowance for the effect of changing levels of background concentrations over time. In addition, the non-linear characteristics of the concentration/forcing functions have been reduced to a linear approximation, which is only valid for small concentration changes. However, despite these issues, given the inherent uncertainties in the science, the model is likely to provide an adequate evaluation of global equilibrium temperature change.

The most important deficiency of the spreadsheet model is the inability to include the effects of the ocean in slowing global temperature change. This has major effects in determining the timescale of temperature change, and therefore is important in the context of the project. The modelling of these effects is too complex to include in a spreadsheet model. Hence, the alternative approach using the STUGE model has also been adopted.

### The STUGE model

The STUGE model (Wigley et al, 1991) provides a more rigorous approach to climate modelling, whilst remaining easy and quick to use on a microcomputer. It incorporates carbon cycle and ocean models, which are good representations of the existing knowledge in these areas. It was developed as part of the scientific assessment of climate change for IPCC (Houghton et al, 1990), and therefore its approach and results have been the subject of international review. Most importantly, it allows the calculation of realised temperature changes in addition to the equilibrium temperature change resulting from any given profile of radiative forcing over time.

Despite the advantages, it must be noted that STUGE was not designed for the calculation of marginal effects or to treat non-continuous emission profiles. There are therefore some important practical problems in running the model to revise the spreadsheet calculations. In particular:
1. The 1990 emission levels are constrained to the values identified by IPCC (Houghton et al, 1990). It is not therefore possible to include incremental emissions in this year;
2. The model is designed to calculate total global effects. Incremental inputs on the scale of a single power plant are negligible on this scale, and therefore the effects are lost completely in rounding errors;
3. The package accepts, as input data, emissions in each decadal year (2000, 2010 etc.). The emissions for intermediate years are calculated within the programme by interpolation. It is not therefore possible to input a step change in emissions corresponding to the start up of a new power station;
4. The results are confined to the period 1990-2100. No information on warming beyond 2100 is available directly from the model.

In this project these problems have been tackled in the following way:

1. The incremental power station is commenced in 2000. This delay of 10 years (compared to the start up date used elsewhere in this work) results in the emissions being injected into an atmosphere with a slightly different set of background concentrations. However, the error this induces in the increments of forcing and temperature change is negligible.

2. The incremental emissions input into the model are equivalent to the output of $10^{14}$ kWh of fuel cycle operation. This level has been chosen empirically to ensure that the output changes are sufficiently large to avoid the rounding errors in the model, whilst not being so large to perturb significantly the operation of carbon or heat sinks. The effects of 1 kWh of operation can therefore be assumed to be equal to $10^{-14}$ of the model run output.

3. The input has been made for a single year (2000), but the interpolation results in a triangular pulse of emissions (from 1990 to 2010, and equal to ten years of the emissions input for 2000). The required data input for 2000 is therefore equivalent to $10^{13}$ kWh of emissions. Because of this form of the input pulse, the radiative forcing and temperature outputs of the model do not correspond to those from an instantaneous emission. However, the discrepancy is only expected to be significant in Year 0 (2000).

4. At present there is no way to use STUGE past 2100. For longer timescales the results of the spreadsheet, described above, are therefore the best available.

The carbon cycle model in STUGE is sensitive to historical levels of carbon dioxide emissions. The incremental atmospheric concentrations, radiative forcing and warming depend upon the background emission levels as well as the incremental emissions. To investigate the effects of background levels, two radically different scenarios have been used:

- The "constant emission" scenario in which the emission levels are maintained at 1990 levels except for CFC11, CFC12 and HCFC22 which are reduced to zero in 2000;
- The "business as usual" scenario, the highest emissions scenario of the IPCC (Houghton et al., 1990).

The difference between these two scenarios, in terms of the incremental realised temperature, was less than 10% for the first 50 years and less than 25% for all of the first 100 years. This discrepancy is relatively small compared to the uncertainties in other aspects of the analysis. To conform to the principles of simplicity agreed for the ExternE methodology, the constant emissions scenario is used.

Analysis of incremental fuel cycle emissions requires two STUGE runs: the first using the constant emissions scenario, the second the same scenario with the incremental emissions added. Subtraction of the first from the second allows calculation of the results of the incremental emissions.

Comparison of results
The results from the spreadsheet model have been compared with those from STUGE for emissions of 880 g/kWh (see Figure 12.1). The curves which are directly comparable are those for the equilibrium temperature rise from the two different models.
Figure 12.1 Comparison of realised and equilibrium temperature rise from the spreadsheet and STUGE model predictions for the coal fuel cycle.

It can be seen that there is agreement to within about 25% throughout the first 100 years, apart from in Year 0, for which the STUGE result is artificially low due to the interpolation procedure described above. With this exception, the agreement in early years is good, but there is a larger divergence of results in the later years, with the spreadsheet model giving rather higher values.

Closer inspection of the results reveals that the difference is due to variation in the predicted levels of atmospheric retention of carbon dioxide. This is not particularly surprising: the spreadsheet carbon cycle model is very crude and makes no allowance for background concentration levels. It is concluded that the STUGE values are likely to be more realistic. However, the difference in results from the two modelling approaches is significantly less than the inherent scientific uncertainties.

The realised warming predicted by STUGE is lower than the equilibrium warming in all years. However, by Year 100, the difference is quite small (13%). This implies that for times longer than 100 years, the equilibrium warming could be used without too much error.

The final warming profile adopted is a combination of the two approaches, using the STUGE output for the first 100 years and the spreadsheet results, normalised to STUGE at t=100 years, for the following 200 years.
12.2.3 Damage estimation

General issues

The impacts of global warming are diverse and potentially very large. They include effects on all of the receptors affected by other forms of pollution described elsewhere in this report. In general, however, the impacts are more uncertain and longer term, and therefore quantification is difficult. The most comprehensive assessments of the impacts, for example by the IPCC, are largely qualitative. To date, the work reported within the ExternE study on global warming has thus been confined to a review of the literature. However, further analysis of individual impacts within an integrated modelling framework is currently being undertaken using the impact pathway methodology (see Section 12.3).

Given the uncertainty in the impacts that may result from global warming, it is clear that accurate external costs estimates are not possible at this stage using any methodology. Indeed, the global and long-term nature of the possible impacts, their interactions and the potential for large-scale social change makes the use of micro-economic cost benefit analysis of limited use (Pearce, 1990; Eyre, 1991). In addition, this type of assessment does not address equity issues, which are clearly an important consideration in policy analysis in this field. Ideally an energy-environment-economy equilibrium model is needed to address the macro-economic aspects of the problem.

However, it is clear that the implications of global warming at unprecedented rates have serious implications for natural ecosystems and for human activities. For the former, at the projected rates of warming, climatically defined vegetation zones would move at speeds faster than forests can naturally migrate. Moreover, montane, island and other physically isolated ecosystems may be unable to migrate. Other potentially affected areas include coastal wetlands, which are at risk from inundation as sea levels rise. It is assumed that coastal protection measures will be concentrated in heavily populated areas, and therefore that many wetland ecosystems will be lost with consequential effects on local fisheries. The overall impacts on biodiversity will also be significant. Attempts to value these impacts encounter the same problems as the impacts of other pollutants on terrestrial ecosystems - at present no credible monetary estimate can be given.

Identifying the main human activities likely to be directly affected by climate change is not too difficult. Primary production (agriculture and forestry) is obviously climate sensitive as are the demand for energy and the supply of water. These are the sectors in which the largest economic impacts are expected, along with coastal protection against sea level rise. Nevertheless, estimation of these impacts is rendered difficult by poor understanding of the regional variation in climatic change.

As with natural ecosystems, the actual valuation of effects on human activities is problematic. This is because of:
- The incompleteness of detailed impacts studies;
- The large uncertainties in the estimates; and
- The likelihood of significant interactions between different impact categories which are usually considered separately (e.g. agriculture and water resources).
The most important uncertainty of the analysis is the scenario of world development over the hundreds of years in which impacts of current emissions are expected - the capacity of society to deal with the changes resulting from climate change will depend critically on the level of social and economic development. For this reason all estimates of climate change damages should be treated with caution, especially where the underlying scenario is not specified.

Literature review

A range of studies have been reviewed, which estimate that the first order impacts from warming levels due to an equivalent doubling of pre-industrial carbon dioxide concentrations, have an expected value of the order of 1-3% of gross world production (e.g. Cline, 1992; Fankhauser, 1993). This apparent convergence needs to be treated with caution, as it is clear that there is significant commonality of assumptions between the studies. More recent work of the same type within the ExternE Project (Tol, 1995) identifies a somewhat greater range.

Climate change has many potential impacts on human health and life span. It is clearly not possible to review them all, and therefore the most important factors must be identified for more detailed analysis. Most analyses of global warming costs estimate damages primarily by economic sector, such as agriculture, forestry, water etc.

The most frequently quoted of these analyses (Cline, 1992) investigates some of these sectors in detail for the USA and then extrapolates the results to the rest of the world. More recent work (for example by Fankhauser, 1993) uses a similar approach but analyses each separate region of the world. In all cases, authors acknowledge that this ‘partial equilibrium’ approach, which neglects the interactions between impacts in different sectors, is not ideal, but is the best which can be currently achieved.

Within these estimates, it is the impacts on the health and welfare of human populations which are the most controversial areas. Indeed, some scenario and analytical assumptions can give external costs for these impacts which are orders of magnitude larger than those quoted above.

The direct effects of climate on mortality and deaths due to natural disasters (storms etc.) are usually included. However, the mortality implications of impacts in other sectors, for example from changes in agriculture and water supply, are often neglected. In these sectors, impacts are valued through changes in price and demand. This approach may be reasonable in an affluent market economy, but is less obviously relevant to much of the developing world. In such areas, crop failure and drought may already have major impacts on mortality rates, and global warming may increase the problem. Mortality impacts may therefore be dominant, at least under some scenarios.

Studies which do include such indirect effects give results which are dramatically different from other economic analyses. For example a study by Hohmeyer and Gärtner (1992) assumes that reduced soil moisture in many regions will lower agricultural yields and that the consequential losses in production will fall mainly on the poorest people in the poorest countries. They postulate additional mortality impacts of 45 million per year from starvation alone. This compares to Fankhauser's estimate of 0.24 million annually from all global
warming related impacts. The difference arises from a methodological dispute which is clearly identified in the literature on the impacts of global warming (e.g. Kane, 1992; Harvey, 1992). For these reasons, the exact approach and assumptions affect the results considerably.

The valuation of mortality poses different, but equally difficult issues. Hohmeyer and Gärtner value each life lost using a statistical value of life (SVL) of $1 million. This is calculated on the basis of willingness to pay (WTP) studies and is of the same order of magnitude as the value of 2.6 MECU used in this study and derived in a similar way.

However, this value is derived from studies in OECD countries. Similar studies have not been undertaken in developing countries, but it is generally assumed that much lower values would be found. Certainly in the type of scenario painted by Hohmeyer and Gärtner, where the world's poorest people are being asked to pay most of the price of global warming, it can be argued that the world's major decision makers do not, in practice, value their lives very highly at all.

The alternative proposed by Fankhauser is to use lower values of SVL for developing countries - $0.1 million is suggested in the absence of any empirical data, compared to $1.5 million in OECD countries. Fankhauser is at pains to point out that this difference should be interpreted only as a WTP rather than any assessment of relative worth of individuals in different countries. Of course, this distinction is important. Indeed, it is axiomatic in the utilitarian philosophy, which underpins welfare economics, that all people are of equal value. The problem of different monetary values for different countries arises solely because WTP does not reflect the changes in utility resulting from the same expenditure in different countries. The issues of valuing global warming are therefore inseparable from wider questions of development and the world economic order.

A rigorous, ethically defensible, cost benefit analysis of global warming would need to assess changes in utility not merely willingness to pay. Such an assessment would be a significant variation from the normal practice of welfare economics and, as such, is outside the scope of this project. Nevertheless, it is urgently required.

It is concluded that, at present there is no consensus on assessment of mortality effects. The result of using Hohmeyer and Gärtner's estimate of increased mortality with a SVL of 2.6 MECU gives a damage cost which is more than 1,500 times larger than studies based on affluent market economies. The differences arise primarily from ethical judgements and scenario assumptions rather than scientific uncertainties. In these circumstances, external cost analysis can play an important but restricted rôle. Decisions are not merely technical, but include the type of ethical and political judgements, not amenable to cost benefit analysis, which should only be made through the democratic process.

12.2.4 Damage assessment

The diverse estimates of damage costs at 2xCO₂ levels can be used to estimate the corresponding external costs of greenhouse gas emissions from fossil fuel cycles. The damage cost (as a fraction of world production) is applied to the annual gross world
production of approximately $2 \times 10^{13} \text{ (Pearce et al, 1992). This gives the annual damage cost of } 2x\text{CO}_2. \text{ These costs are assumed to correspond to } 2.5 \text{ K temperature rise from global warming - the best estimate of } 2x\text{CO}_2 \text{ global mean equilibrium warming.}

The warming per kWh figures derived from the methodology described in 12.2.2 are used to calculate the external costs of global warming from the relevant fuel cycle (assuming the damage function is linear).

This approach involves a number of important assumptions, and the results it produces are particularly sensitive to:

1. The climate sensitivity, assumed to be 2.5 K, but more realistically somewhere in the range 1.5 to 4.5 K for a doubling of CO$_2$ levels, estimated by the IPCC;
2. The damage for a given level of warming (i.e. the % of gross world production);
3. The discount rate. An illustrative range of 0 to 10% is used in this project.

The initial estimates of global warming damage assume linear damage functions at relatively small temperature rises. They neglect the larger impacts likely in developing countries and the possibility of unforeseen catastrophic damages. Such assumptions clearly lead to a tendency to underestimate costs.

### 12.3 The Impact Pathway Methodology

#### 12.3.1 Introduction

The number of potential impact pathways which can be defined for the effects of climate change is enormous. Moreover, the pathways may be strongly inter-dependent, so the impact pathway will only define the elements of an accounting framework; further analysis will be required to assess synergies. At the time of writing, this approach has not been fully implemented within ExternE, and therefore the following sections describe work which is planned and in progress, rather than completed. Inevitably some methodological details may change with the development of practical implementation experience.

The methodology described below is therefore incomplete and inadequate to characterise all the externalities of climate change. Nevertheless, it is more comprehensive, rigorous and consistent with the paradigm of external costs than any of the work on which frequently quoted global warming damage estimates are based. In this sense it is a significant development.

In order to assess the marginal impacts of climate change, it is necessary to describe the elements of an underlying meteorological, demographic, social and economic scenario. Most review estimates of total global damages (corresponding to double pre-industrial carbon dioxide equivalent greenhouse concentrations) do not specify a scenario and, in practice, draw on studies with different underlying assumptions. This is clearly inadequate for a rigorous impact pathway approach.
In the ExternE Project, the scenario which will be studied initially will be the IPCC 92a scenario (broadly business as usual). This is the type of scenario in which climate change damages might be expected to give rise to serious concern. The choice should not be interpreted as any judgement on either the likelihood or desirability of this scenario - both issues are outside the remit of the ExternE Project.

12.3.2 Climate sensitivity to greenhouse gas emissions.

The global climate sensitivity to greenhouse gas emissions is estimated by MAGICC (Model for Assessing Greenhouse Gas Emissions and Climate Change), which is being used by the IPCC for their 1995 assessment. MAGICC combines a set of greenhouse gas cycle models. These convert user-specified input emission scenarios to concentrations, with an upwelling-diffusion-energy balance ocean climate model and a set of ice melt models to determine future time-dependent changes in global-mean temperature and sea level. MAGICC includes the main greenhouse gases; CO₂, sulphur dioxide, nitrous oxides, VOCs, and CFCs. Parameters are specified for:

- Global average climate sensitivity;
- Carbon, temperature and methane feedback;
- Atmospheric residence time of greenhouse gases;
- Forcing due to ozone and sulphur dioxide;
- Ocean diffusion and upwelling;
- Elements of the global ice-water balance.

In the ExternE Project, it is proposed that the default values are used for these components of the model, corresponding to the IPCC analysis best judgement.

High, central and low estimates of global climate change are calculated, based on estimates of global-average equilibrium climate sensitivity (4.5, 2.5, and 1.5 K, respectively). This range of uncertainty is carried through the economic analysis to estimate the spread of potential impacts.

The model output is in the form of global average values of increases in mean annual temperature (the climate sensitivity), global average greenhouse gas concentrations, and sea level rise (without local trends), for every five-year period from 1990 to 2100.

The global climate changes for individual fuel cycles are calculated by running MAGICC as follows:

1. Run IPCC 92a scenario in MAGICC (using default IPCC 92a emissions);
2. Add fuel cycle emissions, scaled up to 1000 fuel cycles;
3. Run MAGICC with combined emissions from IPCC 92a and the fuel cycle;
4. Subtract results of IPCC 92a (1) from the 1000 fuel cycles (3);
5. Divide the results (4) by 1000 to give the contribution of the fuel cycle.

For each fuel cycle of interest, three reference scenarios (low, medium and high estimates from IPCC 92a scenario) are combined with three policy scenarios (low, medium and high).
12.3.3 Regional climate sensitivity

The global climate change for the IPCC 92a reference scenario, as estimated by MAGICC is used to compile a scenario of regional climate change in 2100. The basic steps are:

1. An equilibrium GCM experiment was chosen to capture the spatial patterns of changes in temperature and precipitation. The Goddard Institute for Space Studies (GISS) scenario was selected since it is widely available, has become a standard for impact assessment, and has relatively small changes (i.e. it is conservative in the estimates of impacts).

2. The GISS GCM results are standardised by subtracting the global-average annual temperature change from the monthly temperature change. This results in the spatial distribution of temperature changes above and below the global climate sensitivity (i.e. the global average warming in 2100 in this scenario). Monthly precipitation anomalies are also scaled to the global climate sensitivity.

3. The time-dependent projections of global climate change from MAGICC are then multiplied by the GISS monthly anomaly fields. The result is projected changes in temperature and precipitation for a specific year (2100) and emission scenario (IPCC 92a).

4. The projected changes are added to the baseline climate data to provide a scenario of the future climate. Impact models run for the baseline climate are run again for the new climate to calculate the first-order sensitivity to climate change.

12.3.4 Reference baseline

**Baseline climatology**

Mean monthly temperature and precipitation for the current climate are from an established data base developed by Leemans and Cramer for IIASA and used in the IMAGE model. The data are gridded at a resolution of 0.5° latitude by 0.5° longitude, comprising 259,200 grid cells. This resolution provides regional specificity for the impact analysis, while still being reasonably tractable in terms of computing requirements. The climate data are held in a Geographic Information System, which is also used to calculate regional scenarios and impacts.

**Economic data**

Country level data on population, GNP, and the breakdown of GDP between the categories of agriculture, industry and services are taken from the 1993 World Bank Development Report, which presents data compiled for 1990. Where these data are incomplete (in the case of the GDP breakdown), values taken from the appropriate regional grouping are used.

The IPCC 92a scenario does not provide explicit estimates for the growth of each economic sector for each country to the year 2100. Thus regional projections from the IPCC and other sources are used, and attributed to each country based on broad relationships between GNP and impact indicators or current and estimated shares in regional resource demand.
12.3.5 Detailed impact models

At this stage it is proposed that four sectors should receive detailed analysis:

- Coastal zones;
- Energy demand;
- Water resources;
- Agriculture.

For coastal zones, global average sea level rise is projected from MAGICC.

Energy demand for space heating and cooling is a function of the difference between the ambient temperature and a "base temperature". For example, when temperatures exceed a base temperature it is expected that no energy will be required for space heating. At temperatures below the base temperature, heating demand is proportional to the product of the temperature shortfall and the time for which the shortfall is maintained. For space cooling demand is a function of the extent to which ambient temperature exceeds the base temperature.

The impact measure for modelling the effect of climate change on water resources is precipitation (P) minus potential evapotranspiration (PET). PET is calculated based on the Thornthwaite equation, modified to correct for overestimates at poleward latitudes. The balance of monthly P-PET is accumulated over the year, providing annual estimates of water surpluses (months where P exceeds PET) and water deficits (PET exceeds P). For impacts on water resources, changes in water surpluses (i.e. surface runoff and groundwater recharge) are most important. As for energy demand, the impact is weighted by the geographic area affected. The outcome is an index measuring the percentage change in water deficit for each country.

For agriculture, the most robust framework, readily applied in a spatial model, is to calculate a set of first-order indices that relate to broad resource requirements. The indicators used in this study are based on vegetation models:

- Biotemperature between 5°C and 25°C, where biotemperature is the accumulation of monthly mean temperature over 0°C divided by 12 (the number of months in a year);
- Precipitation greater than 500 mm per year and less than 2500 mm per year.

12.3.6 Damage estimation

Basic approach

The principal studies of climate change damages have taken the approach of identifying the sectors which will be most affected, and estimating the likely impacts separately in each one (Table 12.1). The discrete analysis of impacts in these separate categories may underestimate the effects of interactions between the sectors. Such effects, resulting from long-term changes in economic equilibrium, are extremely difficult to estimate in general, and the studies to date have not attempted to do so.
Table 12.1  Types of impact and associated sectors for valuation.

<table>
<thead>
<tr>
<th>Impacts: Valuation of:</th>
<th>Temperature Rise</th>
<th>Precipitation Change</th>
<th>Sea Level Rise</th>
</tr>
</thead>
<tbody>
<tr>
<td>Coastal Protection</td>
<td></td>
<td></td>
<td>For valuable coastline</td>
</tr>
<tr>
<td>Loss of Wetland</td>
<td></td>
<td></td>
<td>Where coast is protected</td>
</tr>
<tr>
<td>Bio-diversity</td>
<td></td>
<td>Caused by changes of habitat</td>
<td></td>
</tr>
<tr>
<td>Loss of Dryland</td>
<td></td>
<td></td>
<td>Where coast is not protected</td>
</tr>
<tr>
<td>Human Migration</td>
<td></td>
<td>From drought prone areas</td>
<td>From lost dryland</td>
</tr>
<tr>
<td>Agricultural Potential</td>
<td>Change in soil water balance and area suitable for cultivation</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Energy Demand</td>
<td>Heating and cooling</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Water Resources</td>
<td>Change in water balance</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Other Sectors</td>
<td>Includes Multiplier Effects, Extreme Events</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

This project follows the same approach, leaving studies of interactions to future economic models. It begins by estimating the global climate sensitivity to the IPCC 1992a scenario between the years 1990-2100, and then follows through the impacts on each of the important sectors.

**Major damage categories**

In analysis of the effects of climate change, monetary impacts are scaled against the gross national product (GNP). In some cases, the impacts are assumed to change in the future in proportion to GNP. This applies in particular to estimates of agricultural impacts, and the contingent valuations placed upon the use of land or the existence and use of biodiversity. In this respect, the results of the analysis are sensitive to the assumptions about economic growth contained in the IPCC 92a scenario which forms the baseline of the study.

Many of the methods of costing are biased in favour of countries where the per capita income is high. The most obvious example is the valuation of the loss of coastal lands in countries such as Bangladesh or Kiribati, where land prices are low, as against the valuation of low-lying land in Europe. It is widely acknowledged that economic variables do not adequately measure human well-being, and the concentration upon macro-economic measures such as
Global Warming

GNP neglects important distributional issues. The approach taken in this project tries wherever possible to avoid the attribution of widely differing values to human assets in different countries.

Some of the cost categories used in this type of analysis are difficult to monetise - in particular the loss of biodiversity. There is a need to consider other ways of determining the social importance placed upon such impacts. It may be that there is no socially neutral scale of measurement against which the costs and benefits can be assessed. However, at this stage, this project has followed previous studies in making use of contingent valuations based upon willingness-to-pay.

Arguments based upon inter-generational issues would imply a very low, or zero, or even negative rate of discounting - an approach which would render a cost-benefit analysis impossible to implement for the long-term. It has also been argued (e.g. Tol, 1993) that different regions of the world would have a different social rate of time preference, on the grounds of differences in per capita income. The use of such regionally differentiated rates would however be sensitive to assumptions about the rate of per capita income growth over a long time period. The pace of change over the last 50 years suggests that such assumptions would not be robust. Within this project a variety of rates are used to enable assessment of sensitivity to discounting.

Other sectors and disasters

Since climate change necessarily implies changes to a huge range of natural and human activity on Earth, additional impacts must be considered. Drawing upon previous studies, the most significant impacts modelled in detail are reviewed. In most cases it is not possible (or prudent) to assign an economic value. Where possible, a range of estimates are provided from previous assessments and it is noted how they may differ from the assumptions used in this project. For the impact of extreme events and disasters, a first estimate is provided, albeit with a large range of uncertainty.

12.4 Conclusions

In this chapter, two methodologies for estimating impacts resulting from global climate change have been described. To date, the analysis conducted in this project has made no independent contribution to the assessment of global warming damages. The results quoted in the fuel cycle reports (European Commission, 1995a-c) are therefore only the results of literature review and are not to be interpreted as the results of ExternE. Our literature review has identified major problems in the existing literature, which are being tackled by ongoing work using the impact pathway methodology.

It is clear that assessment of these external costs are most dependent on the assumptions made about the trajectory of world social and economic development. All attempts to value these impacts require important normative judgements, and therefore the potential for synthesis or consensus is remote.
12.5 References


13. NOISE

13.1 Introduction

Noise is emitted from almost all stages of all fuel cycles. Despite this fact, it has historically rarely been considered as a major external cost of energy. It is local in nature - with audible impacts rarely extending more than a few kilometres from the source. Noise issues are usually considered in detail at the planning stage. In many, but not all cases, there are abatement measures which can be taken to reduce noise emissions. These are usually specified to reduce nuisance to nearby populations to levels considered acceptable for the local environment.

As far as fuel cycle external costs are concerned, most attention has been paid to the potential noise externalities of extensive renewable energy sources in rural areas, particularly wind turbines, for which some degree of aerodynamic noise is unavoidable. Whilst wind turbines are much quieter than many other energy conversion devices, the absence of many other environmental impacts and the sensitivity of amenity in some potential areas of deployment has focused attention on noise. The methodology described below has therefore been developed primarily for the analysis of wind turbine noise. It is, however, directly applicable, to other energy technologies, but for the other fuel cycles has often been adopted in a simplified form.

Noise levels are generally specified on the decibel, or dB, scale, which is defined in terms of the sound pressure (in Pascals) by the equation:

\[
\text{Noise level} = 20 \log_{10} \left[ \frac{\text{Sound pressure}}{20 \mu Pa} \right]
\]

13.1

This is usually adjusted to allow for the variable sensitivity of the human ear to sound of different frequencies by using the dB(A) scale. There are many variants of this scale to allow for the fluctuations of noise levels over time, notably the \( L_{Aeq} \) scale, which is the level equivalent to the mean sound energy level.

Specific numerical criteria for wind noise are set at a national level in many European countries (Hayes, 1992):

- Denmark has statutory orders which require noise levels (\( L_{Aeq} \)) of:
  - < 45 dB(A) for all neighbouring properties, and
  - < 40 dB(A) in residential areas and other noise sensitive locations;
• The Netherlands has advisory levels ($L_{Aeq}$) for all industrial noise of:
  < 40 dB(A) in rural areas,
  < 45 (day), 40 (evening), 35 (night) dB(A) in quiet residential areas, and
  < 50 (day), 45 (evening) and 40 (night) dB(A) in all residential areas;
• Germany has recommended levels ($L_{Aeq}$) of:
  < 65 (day) and 50 (night) dB(A) in commercial areas,
  < 60 (day) and 45 (night) dB(A) in mixed areas,
  < 55 (day) and 40 (night) dB(A) in general residential areas,
  < 50 (day) and 35 (night) dB(A) in pure residential areas.

In other countries there are statutory and advisory procedures for assessing noise levels, for example in the UK the British Standards for noise issues (BS 4142 (1990); BS 7445 (1991); BS 5228 (1984)). The relevance of these to external costs is as follows:
• BS 4142 defines methods for assessing the level of noise in mixed residential and industrial areas, including the use of the $L_{Aeq}$ measure. It advises that noise caused by a development should be compared with existing background noise levels and that an increase of 5 dB(A) is of ‘marginal significance’. This criterion is widely used for new developments. However, the standard explicitly excludes the advisability of using the criterion where background levels are <30 dB(A) - a level which is typical of rural areas, where some generation sources may be located. The standard also advises the imposition of a 5 dB(A) penalty for noise with a tonal content.
• BS 7445 mainly refers to methods for assessment of environmental noise, including a procedure for defining whether noise has a tonal content.
• BS 5228 refers to noise from construction sites. However, it is relevant to fuel cycles in that it estimates that night time noise levels ‘may need to be as low as 40 to 45 dB(A) to avoid sleep disturbance’, and advises the application of a 10 dB(A) penalty in the assessment of night time noise.

13.2 The Noise Impact Pathway

The noise impact pathway is shown in Figure 13.1. It includes aspects to deal with the dispersion and abatement of sound, tonality, intermittency, time of day, the interaction with background noise and human attitudes to noise.

The analysis at each stage of the impact pathway is dealt with in more detail in the following sections.
There are various types of noise emitted by energy technologies. The mechanisms by which sound energy is produced are complex and frequently only partly understood. As far as the analysis of external costs is concerned the detail of the source and mechanisms is unimportant. Most complaints about noise relate to sources where the noise has a strong tonal component. In addition, the timing and intermittency of the source can affect nuisance. All that is required to specify the source of the external costs is:

- The magnitude of the noise emissions (in dB(A) relative to 1 pW);
- Any variation of noise as a function of time of day;
- An analysis of any tonal content; and
- An analysis of intermittency.

**Figure 13.1** Impact pathway for noise.
In general, there is data available on the sound levels emitted by fuel cycle sources. Where this is not available, the emission level may be calculated from measurements of the noise at known distances from the source using the relationship noise and emission level given in equation 13.6 below. In some cases, the expected observed noise levels at all locations of interest may be available; in these cases estimation of the emission level and the dispersion may be avoided completely. For continuous sounds, it is the $L_{\text{eq}}$ measure which is of interest. For intermittent sounds it is the peak level.

### 13.4 Propagation of Sound

The propagation of sound through the atmosphere is well understood at the theoretical level. Nevertheless, modelling the propagation in real reference environments presents some practical problems.

#### 13.4.1 Spreading

The most simplistic model of noise propagation assumes that sound disperses from a point source with no attenuation or obstruction. This is a reasonable approximation in some conditions, especially if what is sought is the maximum noise at a given point from a single sound source. This approach is used for the coal and gas fuel cycles power station noise and its lack of sophistication is justifiable because of the relative unimportance of noise as an externality in those fuel cycles. In this approximation, the noise level, $L_p$, in dB(A) at a distance, $r$, from a source of power, $L_w$, is given by:

$$L_p = L_w - 10 \log_{10} (2\pi r^2)$$

This is the $L_{\text{eq}}$ value whilst the source is operating. The inverse square law formulation derives from the assumption that the sound disperses uniformly through a hemisphere above flat reflective ground. It leads to a reduction of $20\log_2$ (approximately 6) dB(A) per distance doubling.

For fuel cycles where noise is believed to be a potentially important impact, consideration of more complex approaches is justified.

#### 13.4.2 Air attenuation

A review of different propagation models (Bass, 1992) indicates two broad approaches. The first, based on the work of an IEA expert study group (Ljunggren, 1988) is a refinement of the simplistic model presented above. At distances very much greater than source size, the source may be treated as a point source. The effects of attenuation by the air are added so that:

$$L_p = L_w - 10 \log_{10} (2\pi) - 20 \log_{10} r - \alpha r$$

13.3
where $\alpha$ is the coefficient of absorption of sound in air. The values of this coefficient are well documented as a function of atmospheric variables (temperature, pressure and humidity) and frequency. Under normal atmospheric conditions, $\alpha$ varies from zero at frequencies below 125 Hz to 0.056 dB(A) m$^{-1}$ at 8000 Hz. For the typical broad band frequencies of noise 0.05 dB(A) m$^{-1}$ is a common estimate of the average value. Using this value, the numerical form of equation 13.3 is:

$$L_p = L_w - 8 - 20 \log_{10} r - 0.05r \tag{13.4}$$

### 13.4.3 Other factors

The approach of the IEA model still neglects various factors, notably ground absorption, wind, other meteorological conditions and physical barriers. The qualitative effects of these is as follows:

- Many types of surface, notably soft ground, deviate significantly from the approximation of a perfectly reflecting surface. The effect of ground absorption reduces noise levels away from the source.
- The main effect of wind and temperature gradients is to refract the sound. Where the refraction is away from the ground - up-wind of the source and under normal meteorological lapse conditions - the effect is to increase the attenuation rate above predictions without refraction. However, down-wind of the source and under inversion conditions there is a tendency for sound to be refracted back towards the ground and noise levels can be higher than predicted by spreading and air attenuation alone.
- Acoustically solid barriers - hills, walls etc. - significantly reduce noise levels. The combined effect of these factors is that noise does not vary as a function of distance from the source only, as indicated by equation 13.4. If accurate predictions at individual locations are required then other factors need to be addressed. Models exist which do take into account these factors. These have been surveyed elsewhere (ISVR, 1990; Bass, 1992) and are not described in detail here. The most commonly used predictive tool is the CONCAWE model (Manning, 1981), which is an empirically derived model incorporating the factors described above.

Alternatively it is possible to use ‘ray tracing’ models (ISVR, 1990). These are more complex to run, but are believed to be more appropriate for the calculation of long range effects and the identification of individual locations which may be severely affected.

### 13.4.4 Model comparison and choice

Comparison of the CONCAWE model (Manning, 1981) and the IEA model (Ljunggren, 1988) shows that the former is significantly better (Bass, 1992), modelling measured noise levels in the range 150-350m from the source with an r.m.s. error of 1.9 dB(A). The comparable figure for the IEA model was 4.5 dB(A). The difference results from overestimation of noise by the IEA model due to neglect of ground absorption.
At distances in excess of 2 km from the source, the results of ray tracing and spherical spreading models are significantly different (ISVR, 1990). Field measurements show that the higher noise estimates of the ray tracing approach are more reliable. Essentially spreading type models neglect multiple path effects which are significant at these distances. At this range, the noise tends to attenuate in proportion to $10 \cdot \log_{10} r$, as opposed to $20 \cdot \log_{10} r$ in the near field.

Despite the existence of superior models, the IEA model is used in this analysis. The superior performance of ray tracing models at distances greater than 2 km is of only marginal importance as most of the noise impact of fuel cycle sources is on locations within 2 km (even allowing for larger populations at greater distances). The better performance of the CONCAWE model would make this a better choice for more accurate analysis. However, a vastly increased amount of reference environment data would be required. This would make the task of analysis for different locations much more difficult. The additional error of 3 dB(A) introduced by using the IEA model is believed to be acceptable. Considering that it is a systematic overestimate, it will result in overestimation of the noise externality.

### 13.5 Observed Sound Levels

The IEA model in the form of equation 13.4 is used to calculate the noise impact. The calculations may be done using a spreadsheet, originally designed for wind farms (Edwards, 1993; Gildert, 1993). Some further modifications to allow assessment of longer range impacts have also been included.

The position of each source and observer location is defined on a two dimensional grid. The source positions are taken from the relevant Environmental Statements and the house positions from maps of the areas. The spreadsheet calculates the noise level from each source at each house position. The combined effects at each house are then calculated by summation over all N sources. Bearing in mind the logarithmic scale which defines the dB, this requires use of the following formula:

$$L_{p,\text{total}} = 10 \cdot \log_{10} \sum_{i=1}^{N} 10^{L_{p,i}/10}$$  \hspace{1cm} 13.5$$

A similar approach is used to summing the source noise level, $L_{p,\text{total}}$, and the background noise level, $L_{\text{back}}$. The observed noise level is:

$$L_{\text{obs}} = 10 \cdot \log_{10} \left( 10^{L_{p,\text{tot}}/10} + 10^{L_{\text{back}}/10} \right)$$  \hspace{1cm} 13.6$$

The background noise level may be an important parameter in the analysis. The level chosen should ideally be based on survey data.
Of course, for people inside buildings the noise level is much reduced and the quality of this noise insulation varies. However, the difference between houses is relatively small and the data on internal noise levels is harder to model. Analysis is therefore usually undertaken on the basis of external noise levels, and this approach will be followed here. However, it needs to be noted that this approach avoids some special cases. In particular, there is evidence that residents and visitors in less acoustically insulating accommodation (e.g. caravans) experience significantly higher noise levels (Spode, 1992).

13.6 Perception of the Observed Noise Levels

13.6.1 Problems of perception, amenity and valuation

The observed sound levels calculated using the procedure described above are objective measures of the noise which individual observers will hear. The levels calculated are subject to error for all of the reasons described, but they have a clear physical meaning. However, they are not necessarily a good indicator of the noise which people will perceive, the change of amenity they will experience or the value they will put on that change. These issues are dealt with by the final two stages of the impact pathway set out in Figure 13.1 - the perception and valuation of noise. They are largely the province of social science rather than acoustics.

The use of the dB(A) scale for noise measurement is not completely irrational despite the unwieldy arithmetic of addition which it requires (see Section 13.5). The audio (A) weighting allows for the sensitivity of the human ear to different frequencies of sound, and the logarithmic relationship to sound pressure (equation 13.1) is broadly representative of the response of the human ear to sound. The dB(A) scale therefore is a fairly linear measure of loudness, at least in the sense of the response of human auditory system.

However, we cannot go further than this and deduce that the dB(A) scale is a good linear measure of the level of annoyance associated with noise for a whole variety of reasons. The most obvious is the extent to which the noise is unwanted - the output from one’s own stereo is music, but the output from someone else’s is noise. However, even for noise which is externally generated and entirely unwanted there is a range of factors to consider:

- Perceptions vary as a function of time of day and intermittency;
- Different people have very different sensitivity to noise;
- Audible discrete frequencies (tonality) within the noise increase annoyance;
- Perception may be affected by general attitude to the noise source;
- Change in amenity may be affected by existing amenity, particularly background noise levels; and
- Monetary valuation is affected by socio-economic factors in the area.

For these reasons it is not easy to separate out the final two stages of the impact pathway, and they are considered together in this section.
The effect of these issues is to make calculation of the perception, amenity and monetary values of noise more difficult. In particular, for fuel cycle sources there are problems in using the body of knowledge developed for assessment and control of other noise sources, such as industry, traffic and aircraft. The rural areas in which some sources are located may differ from most of the areas in which other noise problems have been studied, in terms of source type, characteristics of the noise, background noise levels, existing amenity levels and socio-economic status of the population. The transferability of studies of perception, amenity and value of noise from these other areas is therefore fraught with difficulty. Caution is required in the transfer process.

With this caveat, some of the variables listed above can be dealt with by rules which have been developed in the general applications of noise analysis and control. Others are more difficult, as described below.

### 13.6.2 Time of day effects

The noise levels calculated by equations 13.3 to 13.6 represent measures of the $L_{Aeq}$ noise when sources are operating. It is necessary to make some allowance for the fact that noise is more annoying at night time than during the day. The extent of this differential annoyance will obviously vary from person to person. However, it can reasonably be assumed that the standards and approaches which have been adopted in noise control represent a good measure of the average. The discussion in Section 13.1 highlights several standards in which a ‘day to night variation’ is adopted. A difference of 10 dB(A) is typical, for example in BS 5228 and the advisory levels used in the Netherlands.

A 10 dB(A) penalty is therefore adopted for night time noise. This is used to adjust both the turbine and background $L_{Aeq}$ noise levels to define a ‘day-night’ noise level, $L_{dn}$, defined as the time averaged continuous sound level after the addition of 10 dB(A) to sound levels between 10 p.m. and 7 a.m. For a constant noise level, this is calculated as follows:

$$ L_{dn} = 10 \log_{10} \left[ \frac{15}{24} L_{Aeq} + \frac{9}{24} \left( L_{Aeq} + 10 \right) \right] $$

The $L_{dn}$ levels for both the source and background noise are used as inputs to equation 13.6 to calculate the observed ‘day-night’ noise level, $L_{dn,obs}$.

### 13.6.3 Intermittency

A variety of approaches can be used to assessing the effect of intermittency on noise impact. Two are used in this work:

- The noise and number index (NNI); and
- Use of the time averaged noise level.

For occasional loud noises, e.g. aircraft taking off and landing, it is conventional to use a ‘noise and number index (NNI)’, which is a function of peak noise level and frequencies. This approach was developed in the context of aircraft movement and is used in ExternE in the similar context of train noise.
Noise

\[ \text{NNI} = P \ (\text{dB}) + 15 \log n - 80 \]

where \( P \) is the peak noise level and \( n \) is the daily frequency.

The peak noise experienced near a railway corresponds to an individual train, and therefore will not be affected by an increased number of trains. The extent of the reference environment affected is defined by the NNI contour at which there is judged to be no measurable nuisance value. Drawing on the work of the Roskill Commission into the proposed third London Airport, it is concluded that this limit is at NNI=30.

However, this technique is not appropriate for quieter noise of longer duration, for which it is normal to use the \( L_{\text{eq}} \) value. To calculate this value for a fuel cycle source, it is necessary to average the noise whilst the sources are in operation over the whole of the year. If the fraction of the year in which there is operation is \( f \), the annual equivalent noise due to the turbines and background noise, \( L_{\text{year,obs}} \), is given by:

\[
L_{\text{year,obs}} = 10 \log_{10} \left[ f \cdot 10^{L_{dn,obs}/10} + (1-f) \cdot 10^{L_{dn,back}/10} \right]
\]

The difference between this value, \( L_{\text{year,obs}} \), and the expected noise without the turbines, \( L_{dn,back} \), is used to estimate the noise disamenity due to the fuel cycle source.

13.6.4 Noise sensitivity

It is well established that some people are more sensitive to noise than others, in terms of their propensity to report annoyance at a given noise level. The usual measure of annoyance is the proportion of people reporting themselves to be highly annoyed either in social surveys or by actual complaints to the relevant authorities. On the basis of empirical work on transportation noise and some theoretical considerations, a functional form for the probability of being highly annoyed \( P(\text{HA}) \) has been proposed as follows (Fidell, 1991):

\[
P(\text{HA}) = \exp \left[ -10^{p/10(D-L_{de})} \right]
\]

where \( D \) is the noise level at which \( e^{-1} \) (37%) are highly annoyed by the noise, and \( p \) is an elasticity which determines the spread of sensitivity to noise. Both are empirical values and could, in principle, vary from place to place for the reasons given above. Typical values obtained in traffic noise studies are \( D=70-75 \) dB(A) and \( p=0.3 \).

In principle, other measures than ‘highly annoyed’ could be used to assess noise amenity. It is certainly far from obvious that this is the best measure for relatively quiet sources. Reporting annoyance presumably corresponds well with the concept of ‘noise nuisance’ used in pollution control, rather than the concept of amenity which is more closely related to utility and value. However, the existing literature is formulated in terms of high annoyance levels, and therefore, this parameter must be used if quantitative results are to be obtained using this approach.
The functional form of equation 13.10 introduces a non-linearity (with respect to dB(A)) into the assessment, but allows analysis of noise on a probabilistic basis.

13.6.5 Tonality

Although the definition of the dB(A) allows for the sensitivity of the ear to sound of different frequencies it does not allow for the undesirability of discrete frequencies or tones in the noise. It is well established that there is an additional annoyance if noise has a strong tonal content. This is allowed for in BS 4142 by the addition of a 5 dB(A) penalty.

13.6.6 Attitudes to the source of noise

The extent to which attitudes to noise differ between sources is difficult to establish. Subjective evidence from objectors to the wind farm at Penrhynddlan and LlidiartysWAUN indicates some antipathy to noise may relate to the fact that residents were given ‘assurances from the developers that the site would be inaudible’ (Lord-Smith, 1993). However, the same objector asserts that the noise is ‘similar to a busy main road’, implying that all noise sources are judged on the same basis.

In conclusion, there seems to be no strong evidence that the same level of noise from different sources has different amenity effects as a result of attitudes to the source of noise.

13.6.7 Effects of background noise levels

There is some evidence that opposition to noise in some locations derives, at least in part, from the fact that low background noise levels are considered an important component of the amenity of the area (e.g. Lord-Smith, 1993). It is not, however, possible to identify or quantify this effect with any certainty.

However, it should be noted that one of the methods which is used below to value the noise from wind turbines (‘dB(A) costing’), in practice makes a considerable allowance for a larger effect at lower background noise levels. Because of the mechanics of arithmetic of noise on the dB(A) scale (see equation 13.6), a given turbine noise will increase the observed noise level more at lower background noise levels. The difference can be very significant as shown in Table 13.1

The incremental noise caused by source noise of 35 dB(A) varies by more than two orders of magnitude as the background noise varies over a reasonable range for a rural location of 20 to 50 dB(A). This difference will feed through directly into the monetary value of the amenity if the ‘dB(A) costing’ method is used (see Section 13.8), as this values the incremental noise. In effect, therefore the ‘dB(A) costing’ method allows for a strong sensitivity to background noise.
Table 13.1  Effect of background noise on incremental observed noise.

<table>
<thead>
<tr>
<th>Source noise, dB(A)</th>
<th>Background noise, dB(A)</th>
<th>Observed noise, dB(A)</th>
<th>Incremental noise, dB(A)</th>
</tr>
</thead>
<tbody>
<tr>
<td>35</td>
<td>20</td>
<td>35.1</td>
<td>15.1</td>
</tr>
<tr>
<td>35</td>
<td>25</td>
<td>35.4</td>
<td>10.4</td>
</tr>
<tr>
<td>35</td>
<td>30</td>
<td>36.2</td>
<td>6.2</td>
</tr>
<tr>
<td>35</td>
<td>35</td>
<td>38.0</td>
<td>3.0</td>
</tr>
<tr>
<td>35</td>
<td>40</td>
<td>41.2</td>
<td>1.2</td>
</tr>
<tr>
<td>35</td>
<td>45</td>
<td>45.4</td>
<td>0.4</td>
</tr>
<tr>
<td>35</td>
<td>50</td>
<td>50.1</td>
<td>0.1</td>
</tr>
</tbody>
</table>

13.6.8 Socio-economic effects

The effects of social class and other socio-economic characteristics of the nearby population on attitudes to noise are not well known. There is a common belief that amenity is more valued in higher income groups. Also, older people, although in general having poorer hearing, may be more sensitive to sleep disturbance. It is not clear if or how these factors should be taken into account. Where, as here, valuation relies on hedonic pricing, it is often assumed additional amenities will be captured in the house prices used. However, it is very possible that amenity factors such as ‘peace and quiet’ make a higher contribution to rural amenity values, and therefore that the transference of noise hedonic pricing between urban and rural situations is subject to error.

13.7 Valuation of the Noise Impact

In this work a weighted average of European studies on noise values has been used. This is based on recommendations of an expert group which has surveyed the European valuation literature (see Part II of this report, regarding economic valuation of impacts). A noise depreciation sensitivity index, NDSI, is defined using hedonic pricing to estimate the depreciation in house prices as a function of ambient noise level. A best estimate is taken of 0.9% depreciation in property prices per dB(A) $L_{Aeq}$.

If this ‘dB(A) costing approach’ is used, most of the perception issues raised in the previous section are avoided. The observed noise levels can be corrected to allow for time of day (equation 13.7) and intermittency (equation 13.8), and the resulting value, $L_{Y_{pra,obs}}$, is used directly as an input to valuation.

The NDSI recommended above was based on studies considering road traffic noise. As noted in the previous section, there are a large number of ways in which both the type of noise and the nature of the reference environment may vary. The transferability of this value is therefore a contentious issue. In particular, there is concern that road traffic studies for noise...
levels in the range 55-65 dB(A) may not be appropriate to use for valuing noise at much lower levels (Gildert, 1993).

An alternative ‘annoyance costing approach’ has been proposed (Gildert, 1993). This uses an approach based on the probability of being highly annoyed (equation 13.9). This probability is taken as a proxy for the loss of amenity at any given location. The value of annoyance is deduced from the same hedonic pricing studies at road traffic noise levels and then applied to the lower noise levels typical of wind farm locations.

The ‘annoyance costing approach’ has the advantage of not using directly the hedonic prices outside the range in which they were derived. On the other hand, it still fails to tackle some of the other transferability issues (notably sensitivity of amenity to background noise), and it uses ‘high annoyance’ as a proxy for amenity, which is intuitively worrying at low noise levels. Both methods are discussed below.

13.7.1 Valuation by dB(A) costing

Conceptually this approach to valuing the observed noise is straightforward. At each house affected by the noise the increment in the yearly average noise is valued using the NDSI. The annual value of noise, AVN, due to the source is then:

\[
AVN = \sum_{\text{all positions}} (L_{\text{year,obs}} - L_{\text{dn,back}}) \times N_{\text{houses}} \times A(P) \times NDSI ,
\]

where N is the number of houses at that location and A(P) is the annuitised average house price.

At these low noise increments, the new noise source is likely to be indistinguishable from the background, and therefore there is a case for arguing that there is no externality. However, this is not the approach which has been taken for small impacts of other fuel cycles, where even indistinguishably small impacts have been counted (e.g. for pollution impacts on crops). Even a very small noise increment is assumed to have some amenity cost.

The implications of this approach are that it is conceivable that most of the impacts could be experienced where they are apparently imperceptible. The conclusion, although counter intuitive, is exactly analogous to the results found for health impacts of emissions from fossil fuel power stations. However, the physics of noise propagation makes such a result very unlikely and the cumulative impact is expected to converge rapidly with distance. For example, even a city of 10 million inhabitants at this distance would suffer negligible aggregate disamenity. In other words noise impacts are truly local, and there is no significant long range effect.

The sensitivity of observed noise to background noise levels was noted in section 13.6.1. The sensitivity of the results to the low estimates for background noise has been investigated in work on the wind fuel cycle, where it is shown it may be significant. This sensitivity needs to be examined to determine the upper range of plausible external costs of noise.
13.7.2 Valuation by annoyance costing

The costing of annoyance is based on the use of equation 13.10. In this study a value of $p=0.3$ is used, although it has been shown that the overall results are insensitive to reasonable choices of this parameter (Gildert, 1993). The value of $D$ is a more important choice. Typical values chosen in transport noise studies are in the range 70-75. It is reasonable to assume that the probability of being highly annoyed at rural background levels is very small, and a probability of $10^{-3}$ has been suggested (Gildert, 1993), resulting in a value of $D=71.4$.

Studies on annoyance and hedonic prices at traffic noise levels are assumed to be compatible and undertaken at noise levels in the range 55-65 dB(A). Using a value of $D=75$ and equation 13.10, it can be shown that $P(\text{HA})$ rises from 7% to 27% over this range, i.e. a 20% increase over the relevant 10 dB(A). Assuming a NDSI value of 1%, the same noise increment which produces a 2% increase in the probability of being highly annoyed also reduces hedonic prices by 1%. It is therefore concluded that a conversion factor of 0.5 may be used from $P(\text{HA})$ to hedonic price depreciation (Gildert, 1993).

Using equation 13.10 on the noise levels from the source, the $P(\text{HA})$ at receptors may be calculated and converted to hedonic price changes.

The results obtained are significantly different from the dB(A) costing method in quiet areas. Even at the locations most affected by the new source, the probability of being highly annoyed is very low, and therefore the disamenity is small.

13.7.3 Comparison between the valuation methods

The values calculated by the annoyance costing method are, by design, similar to those from the dB(A) costing methodology at typical urban noise levels, but are up to two orders of magnitude lower under rural conditions. The values calculated by annoyance costing are increased if lower background noise levels are assumed and the criterion for setting $D$ (0.1% highly annoyed at background) is maintained. However, even a background noise level of 25 dB(A) does not increase the values calculated to the same order of magnitude as the dB(A) costing method. At this background noise level $D<60$ dB(A), implying that a very large fraction of the population would be highly annoyed by noise levels commonly experienced in urban areas. In short, the discrepancy between the two approaches seems to be fundamental to the method, rather than a function of the parameters used.

The reliance of the annoyance method on high levels of annoyance as the proxy for amenity seems to be the cause of the difference. Clearly some people are concerned about the changes in noise levels they experience. Yet the annoyance approach estimates very low probabilities of annoyance, so low that at rural population densities it is unlikely any single individual would be ‘highly annoyed’ by a wind farm. Hence to rely on this as a proxy for amenity seems unwise.
The assumption of the annoyance method - that the probability of annoyance as a function of noise will be the same in rural and urban areas - is likely to be invalid, yet there is no firm data to allow any other quantitative function in the approach. The dB(A) costing method effectively increases sensitivity to new noise sources at low background levels because of the arithmetic of dB(A) addition explained in Section 13.5. But in the annoyance costing approach, this effect is more than offset by the exponential form of \( P(HA) \) (equation 13.10). Thus, if amenity is more affected by a given new source when background noise levels are low, the dB(A) scale may be a better proxy than annoyance for amenity.

It is concluded that the dB(A) costing approach is preferable to the annoyance method for the following reasons:

- It is consistent with valuations of other noise sources using hedonic pricing;
- It does not rely on very low annoyance probabilities as a proxy for amenity;
- It allows for amenity to have greater sensitivity to new sources of noise if background noise levels are low;
- It produces a higher value, more likely to represent an upper limit of the externality value, which is probably of more interest to policy makers.

However, the problems of transferring hedonic pricing studies to rural, low noise reference environments cannot be escaped. The annoyance costing approach cannot therefore be ruled out and may be used to estimate a lower limit of disamenity.

The best estimate of noise disamenity is therefore based on dB(A) costing with a background noise based on survey data, the upper limit on the same methods with a lower limit background, and the lower limit on the annoyance costing method.

### 13.7.4 Valuation of NNI

To calculate the noise nuisance impacts of the levels shown above, it is necessary to know the number of households, \( N \), affected at each NNI level exceeding 30. The noise impact (in household NNI units) is calculated by multiplying the NNI increment by the number of houses affected.

Most of the relevant valuation studies are based on aircraft noise around large airports and again use hedonic pricing. Meta-analysis of aircraft studies indicates a value of 0.45% change in residential property values in affected areas (i.e. above NNI=30). There may be problems of transferability to rail noise, but studies are not available to assess this, and therefore this NDSI is used to monetarise NNI.

Using equation 13.9, the annual value of noise, \( AVN \), in this case is then given by:

\[
AVN = \sum_{NMI > 30} [P + 15.\log(n - 80)] \times N_{\text{houses}} \times A(P) \times NDSI,
\]

where \( N \) is the number of houses at that location and \( A(P) \) is the annuitised average house price.
13.8 Internalisation of the Noise Externality

At first sight it might appear that the disamenity is necessarily an externality. The noise impacts have most of the attributes of the classical pollution externalities:

- They fall outside the site boundary;
- They affect the population in general rather than any specific group; and
- There is no requirement for the payment of compensation.

However, there may be circumstances in which the disamenity is, at least in part, internalised.

Unlike the impacts of pollution of the air and watercourses, the range of the problem is confined to the immediate environs of the plant. The affected population is therefore aware of the existence of the disamenity and can usually identify the source as the unique cause. If property rights to ‘peace and quiet’ exist, the conditions for internalisation through direct bargaining between polluter and polluted might be argued to exist (e.g. Coase, 1960). In practice, such property rights would have to rely on the sanction of prosecution, and therefore they are unlikely to be achieved.

In some special cases, owners of properties suffering noise impacts may be in a better position to bargain. Where properties are adjacent to the source, there may be the scope for controlling access to the site. Payments are sometimes made by developers in these cases. Whilst the payments are notionally for access rights, the possibility that they include some element of a bribe related to future disamenity cannot be excluded.

In general, however, individuals affected by noise are too numerous and insufficiently powerful to act effectively alone. Affected communities are more effective bargainers acting collectively. Where the source is wholly or partly owned by the local community (e.g. many Danish wind farms), it would be surprising if attitudes to noise disamenity were unchanged.

In conclusion, there are some mechanisms which might internalise noise disamenity in some cases. However, the extent of the internalisation cannot be measured, and the conditions are not generally valid. The damage costs calculated are therefore the best estimates of the external costs of noise disamenity from wind turbines.
13.9 References


ISVR (1990) The Prediction of Propagation of Noise from Wind Turbines with regard to Community Disturbance. ISVR Consultancy Services (J.N.Pinder, M.A.Price and M.G.Smith) ETSU WN 5066.


14. VISUAL AMENITY

14.1 Introduction

The impact of fuel cycle facilities on visual amenity is a factor in all fuel cycles. In the context of external costs studies, it has only been recognised as being of importance where analyses have been undertaken for renewable technologies in rural areas. However, it is clear that there are external costs even in the fossil and nuclear fuel cycles - it is just that the greater importance of health and environmental impacts due to emissions usually limits the effort devoted to consideration of visual amenity effects.

For these reasons, visual amenity has only been identified as a priority impact in the low pollution renewable energy fuel cycles. The methodologies considered below have been derived in the context of implementations of the wind and hydropower fuel cycles. Nevertheless, the principles are applicable to all fuel cycles.

Visual amenity is a local scale impact. Because of the heterogeneous nature of landscape the visual effects of the same technology in different places can be expected to be very different. The importance of local variation is increased by the great importance attached to some rural landscapes. The nature and strength of this valuation is clearly a matter of considerable complexity. In the context of external costs studies, however, the ‘value of landscape’ is confined to its neo-classical economic definition - willingness to pay (WTP) for landscape preservation.

The ExternE Project uses the impact pathway approach; the change to the landscape due to a development is assessed and coupled to a monetary valuation. The WTP principle has been followed for valuation. The principle can be implemented in two very different ways:
- WTP for visual amenity preservation is measured directly for a specific development at a specific site using the contingent valuation method (CVM); and
- WTP is based on ‘benefit transfer’ - that is using the results of monetary valuation exercises (usually CVM) at other sites.

The former approach has the advantage of measuring directly the quantity required, subject, of course, to all the problems and potential biases of the CVM. However, it can only be used for sites where actual CVM surveys have been undertaken. In addition, the WTP for visual amenity will usually be bundled with other aspects of amenity - recreational, nature conservation, etc. - so that it is not readily transferable. The latter approach enables damage estimation for other developments and other sites, and therefore is a more generalisable technique. However it suffers from the problems of applying the results of valuations studies away from their original conditions.
Contingent valuation is used in both approaches. The underlying theory, problems, techniques and recent advances are described in more detail in the Economic Valuation part of this report and in the wind and hydro fuel cycle implementation reports (European Commission, 1995). The remainder of this chapter concentrates only on the application of the CVM technique to the valuation of visual amenity by the two above approaches.

### 14.2 Direct Valuation by Contingent Valuation

Direct application of CVM for valuation of amenity changes can only be undertaken for fuel cycle activities where CVM studies have been undertaken. The methodology of the ExternE Project did not allow for undertaking of basic research, such as contingent valuation surveys. The exception to this is the Norwegian hydropower implementation of the ExternE project, which, thanks to the funding by the Norwegian Ministry of the Environment, undertook a CVM study for the site under consideration.

For a new fuel cycle facility, the individuals in that area are offered a package of both positive and negative impacts. These impacts would be experienced simultaneously, to a large extent, with one or another impact reinforcing or mitigating the effects of others in the individuals’ valuation function. Our task is to estimate damage, i.e. to determine the aggregate WTP of these individuals to avoid or obtain this package of impacts. In an impact-pathway damage function approach we estimate and value impacts for each impact pathway (while accounting to the extent possible for areas of double-counting), and then sum over all individuals and all pathways.

An ideal study would be a ‘perfectly’ designed CVM study that asked the affected individuals to state their WTP to avoid (or obtain) the complete package of impacts from a fuel cycle. Any interdependencies would then, in theory, be taken into account in their WTP responses. By conducting a new direct valuation study we also avoid the uncertainty added by the benefit transfer approach used in other methodologies (see below).

The Norwegian hydropower CVM survey was designed to cover a range of impacts: on recreational activities, cultural objects and terrestrial and aquatic ecosystems of the specific development. These impacts include any changes to visual amenity experienced by the local community, but are not limited to visual amenity impacts. By its nature the CVM survey identifies WTP for a wider package of changes.

The Norwegian hydropower CVM study was designed to take into account the procedures recommended in Part II of this report on the economic valuation of impacts. For further details of the CVM study implemented in Norway, the hydropower fuel cycle study (European Commission, 1995) should be consulted.

In assessing the aggregate damages of any scheme, it is not only necessary to measure the individual WTP, but also to determine the extent of the affected population. This problem is common to both valuation approaches and is addressed in more detail in the next section.
14.3 Visual Amenity Impact Pathway

14.3.1 General approach

Using this approach, unlike the direct use of CVM, the impacts on visual amenity can be separated from other effects. As might be expected the impact pathway for visual intrusion is, in essence, rather straightforward. It has considerable similarities to the impact pathway for noise shown in Figure 13.1, in that the burden causes an objective impact on the observer, which in principle is rather easy to calculate, but the perception and consequent valuation of that impact is more complex. The pathway is shown in Figure 14.1.

![Impact pathway for visual intrusion.](attachment:figure_14.1.png)

**Figure 14.1** Impact pathway for visual intrusion.
The ‘visual burden’ is the size, shape and form of the fuel cycle facility. The ‘objective impact’ of this is the visual image to observers in line of sight. This depends on the land form and visibility, which define the ‘zone of visual intrusion’ and the number of observers in that zone. In this context observers may be residents, workers, through travellers or visitors.

The ‘perceived impact’ will depend on attitudes to the existing land form and scenery, the changes to these due to the introduction of the fuel cycle, and on more general attitudes to the energy source. The valuation of the impact will depend on the same social attitudes and other socio-economic factors.

Figure 14.1 attempts to show the range of factors which need to be taken into account in an idealised model of visual amenity externalities. In practice the data and models do not exist to allow all these influences to be considered. The following sections describe what is available and how it is used in ExternE. But first it is necessary to consider some methodological issues that constrain what can be achieved.

**14.3.2 General methodological issues**

To assess the external damages to visual amenity it is necessary, at an early stage, to define the population affected by the visual intrusion. At first sight the issue might be taken to be similar to the amenity effects of noise, discussed in Chapter 12, where it was concluded that the amenity impact was largely confined to the resident population. However, for visual intrusion, this cannot be assumed, at least in areas of scenic importance for tourism. Direct evidence that landscape changes affect visitor welfare is provided by contingent valuation method (CVM) studies (for example Willis, 1991; Hanley, 1993). In these studies, willingness to pay for the preservation of landscape characteristics is made explicit.

The population affected can conveniently be split into four groups:

- Residents in the area in which the wind farm is visible;
- Visitors to the area;
- People who work in the same area; and
- Through travellers.

For the reasons given above, the first two groups at least need to be considered. The third group can usually be neglected for the purposes of this study, as there is likely to be a strong overlap with the residents category. Through travellers may be significant where a facility is close to a major road or rail link.

CVM is the only available technique for evaluation of visual amenity damage. The hedonic price method, used for the valuation of noise impacts, is unsatisfactory because it only captures changes in welfare of resident households. As an alternative, the travel cost method is frequently used in studies of recreational benefits (e.g. Hanley, 1989). However, this does not allow discrimination between landscape value and other recreational benefits, and therefore is not appropriate for the purposes of this methodology. CVM, on the other hand, can address the welfare of non-residents and be tailored to any category of benefit.
Because of problems in the application of CVM, its use for valuation of visual amenity impacts is only applicable to well used and clearly defined areas with visual amenity benefits, but not smaller areas (such as the zone of visual intrusion of a single facility), due to the possibility of substitution with very similar resources. Expressed in less technical language, CVM is unreliable for a single facility because respondents to willingness to pay questionnaires tend to overstate the value visual amenity in a small area, as they are unused to the problem of budgeting for visual amenity. Furthermore, the monetary values derived by CVM will be widely viewed as an inadequate basis for decision making.

Despite the objective of the study to consider marginal damages (i.e. the impacts of a fuel cycle increment), the relevant valuation technique may only be useful to monetise the visual impact over a whole landscape rather than a small area. For the sake of transparency the impact pathway therefore needs to be implemented as far as possible at the scale of the individual facility, but valuation may only be possible over a wider area.

14.3.3 The visual burden

The visual burden is the visible attributes of the facility. Ideally, a range of factors should be considered, including size and design features such as shape, structure, colour and motion. However, in practice there is little information on the effect that most of these parameters have on the visual impact.

Size is the most important consideration because it determines the range of visibility (in clear conditions) and the magnitude of the change in image for an observer at any given location. Colour may also affect visibility as well as having aesthetic implications.

14.3.4 The zone of visual intrusion and objective impact

The zone of visual intrusion (ZVI) is defined as the area of ground from which any part of the facility is visible. The ZVI can be calculated, either manually or by computer, from a knowledge of the local topography. The maximum range of the ZVI is a question not open to accurate definition. The maximum distance from which a facility can be seen depends on the weather and the observer as much as its size.

In practice the ZVI at all distances is significantly affected by the land form. In all cases, vegetation and buildings will further reduce visibility below the levels predicted from purely topographical studies.

Clearly the scale of the visual impact is dependent upon the distance of the facility from the observer. Except in the crudest sense of the angle subtended at the eye, it is not possible to quantify this.

14.3.5 Landscape assessment

It is inevitable that subjectivity is involved in landscape evaluation. However, there are concepts and procedures which can allow the basic features of the landscape, and changes to it, to be described objectively. The most commonly used descriptive procedures in the UK are
those defined by the statutory body with responsibility for rural amenity protection, the Countryside Commission (Countryside Commission, 1987; Countryside Commission, 1993). These distinguish between objective techniques of landscape classification and the more subjective issues involved in landscape evaluation.

Landscape classification may use either a top-down or bottom-up perspective. The former uses desk based studies of the topographic features of the region to outline the general types of land form. This may be the most helpful approach in the development of strategic planning policies and estimation of impact of whole programmes.

Bottom-up analysis, on the other hand, uses field studies to classify landscape in individual locations. This type of analysis is necessary to describe the visual impact of an individual facility. In order to maximise the degree of objectivity, this approach uses a structured approach even to aesthetic factors. A typical classification is set out in Table 14.1

Table 14.1 A classification of aesthetic factors in landscape assessment (Countryside Commission, 1993).

<table>
<thead>
<tr>
<th>Category</th>
<th>Descriptors</th>
</tr>
</thead>
<tbody>
<tr>
<td>Balance</td>
<td>Harmonious</td>
</tr>
<tr>
<td>Scale</td>
<td>Intimate</td>
</tr>
<tr>
<td>Enclosure</td>
<td>Confined</td>
</tr>
<tr>
<td>Texture</td>
<td>Smooth</td>
</tr>
<tr>
<td>Colour</td>
<td>Monochrome</td>
</tr>
<tr>
<td>Diversity</td>
<td>Uniform</td>
</tr>
<tr>
<td>Unity</td>
<td>Unified</td>
</tr>
<tr>
<td>Form</td>
<td>Straight</td>
</tr>
</tbody>
</table>

This formulation cannot, of course, replace aesthetics. Nevertheless, in the hands of experienced surveyors it allows many landscape characteristics to be conveyed in writing. In general, the impact of fuel cycle facilities will be greater if the landscape, before change, is described in terms of attributes towards the left hand column of the table.

The major objective of evaluation in general is concerned with determining the appropriate designation of the landscape. There is no quantitative content to the recommended evaluation process. Given the status of public attitude and value surveys on landscape described below this is not surprising. It must be remembered that landscape assessment is usually undertaken within a context, where the axioms and techniques of welfare economics are not the dominant paradigms.

14.3.6 Perceived visual impact

The perceived visual impact is due to the psychological effect on the individual of the observed change in landscape. Generally, this depends upon three additional factors:

- Attitudes to scenery and natural beauty;
- The existing level of visual amenity;
• General attitudes to the energy source.

Perceptions of scenery and the value placed upon it clearly varies as a function of the aesthetics of the observer. These may be expected to be affected by a range of socio-economic (as well as purely personal) characteristics. However, surveys undertaken to ascertain public attitudes or values seek to measure some average value. In general public choice type surveys are interested in median values, whereas CVM studies of welfare change are more interested in mean values (Willis and Garrod, 1992). In the latter case, the distribution about the mean may be of interest, but does not affect the overall welfare change. The question of variety of individual perceptions can therefore be neglected for the purposes of this analysis.

Direct measures of public attitudes to landscape amenity in different areas are difficult to obtain. In general, both attitude surveys and CVM studies have been concentrated on areas of high recreational and scenic values, exactly the areas in which many technologies are least likely to be deployed.

Human responses to visual images cannot be treated entirely as an aesthetic question. Psychological responses could well be influenced by the perceived function of the object. Attitudes to energy will therefore affect the perception of visual impact. In practice, attitudes may well be affected by the economic and social relationship between the energy source and the community.

14.3.7 Monetary valuation of visual amenity damage

There are few valuation studies directly relevant to specific visual amenity changes due to fuel cycle facilities. Valuation therefore has to rely on the transfer of values from other types of visual impact. The following UK studies have been identified and reviewed for their relevance to the objectives of this work:
• ‘Landscape Values: A Contingent Valuation Approach and Case Study of the Yorkshire Dales National Park’ (Willis and Garrod, 1991) and the related paper ‘Assessing the Value of Future Landscape’ (Willis and Garrod, 1992);
• ‘Valuing Rural Recreation Benefits: An Empirical Comparison of Two Approaches’ (Hanley, 1989);
• ‘The Contingent Valuation of Forest Characteristics: Two Experiments’ (Hanley and Ruffell, 1993).

These are reviewed in more detail in the Wind Fuel Cycle Study (European Commission, 1995). More studies, such as those undertaken in the Norwegian hydropower study (European Commission, 1995), but specifically restricted to visual impacts, are required if a database of values suitable for meta-analysis is to be established. In the meantime, only approximate values for a restricted range of landscape types are available.
14.3.8 Aggregation

In ExternE it is assumed that the usual (and simplest) aggregation procedure can be used:

\[
\text{Aggregate damage} = \text{Mean WTP} \times \text{Affected population}
\]

14.1

The practical difficulty for energy sector developments lies in the definition of the affected population. For the reasons given above, it is believed the assessment can be confined to two major groups - residents and tourists. This is consistent with the approach taken in the most relevant valuation study (Willis and Garrod, 1991). The total damage is therefore of the form:

\[
\text{Damage} = \left( N_R \times WTP_R \right) + \left( N_T \times WTP_T \right),
\]

14.2

where \(N\) is the number affected and \(R\) and \(T\) represents residents and tourists respectively.

To avoid embedding problems, the resource chosen for valuation should ideally be a well defined landscape region. However, such a choice is necessarily arbitrary to some degree, and therefore it may be helpful to express equation 14.2 on a per unit area basis:

\[
\text{Damage / Unit Area} = \left( N'_R \times WTP'_R \right) + \left( N'_T \times WTP'_T \right),
\]

14.3

where \(N'\) is the number of people affected per unit area.

For any given area, calculation of the resident population is relatively straightforward. However, in some areas of natural beauty the tourist population is typically very much larger, and therefore the tourist number is crucial. Outside these areas, the density of residents may be higher, but the density of tourists much lower.

14.3.9 Internalisation of visual amenity damage

Visual amenity is a classic public good. In most land use planning systems there is no obvious scope for payment of compensation for loss of visual amenity over the ranges relevant to wind turbines. With the exception of any monies paid by the developer for tree planting or other screening efforts, there is therefore no internalisation. The damages calculated are therefore externalities.
14.4 References


15 MAJOR ACCIDENT ASSESSMENT

15.1 Introduction

In general, a major, or severe, accident is defined as an event that leads to the death of 10 or more people. For technological hazards caused by man, about 60% of all such mortalities arise from transportation. The second largest cause is energy production (Fritzsche, 1992), accounting for some 25% of deaths from severe accident world-wide from 1970 to 1985. This figure is likely to be an overestimate as it is dominated by one single incident - the death of 15,000 people in India due to the overtopping of a dam. Nonetheless, it clearly demonstrates the importance of assessing potential major accidents arising in the energy production sector, within the external costs framework.

Within the ExternE Project, a risk based approach has been used to evaluate severe accidents. Ideally, such assessments require good statistical data to estimate both the probability and consequences of a severe accident. Whilst such data is available for some cases, for example in some occupational activities, in general there are many difficulties in applying this method consistently.

There have been several attempts to collate data for severe accidents arising from energy production. The use of such studies as a means to predict current potential accident rates raises a number of problems. The first lies in the definition and reporting of severe accidents; statistical records from different countries may be inconsistent or poorly categorised. A further problem lies in the continuing evolution of technological and safety practices over time; in many cases, even where adequate data exists, historical accidents are no longer representative of facilities or working practices in operation today, for example with historical coal mine disasters. Finally, when relying on historical data, some care must be taken in translating data from technologies or social influences which are not relevant to the technology in question. For example, new large scale hydro developments are now rare in most European countries. Similarly, in the case of nuclear power, the technology and safety practices that led to the major reactor accident at Chernobyl and the explosion at the waste site in the Ural Mountains cannot be considered representative of current EU technology and practices.

The issue of severe accidents is potentially most important with nuclear energy, because of the potential consequences and the long time frames over which impacts may persist. In response to this, probabilistic safety assessment (PSA) techniques have been developed to aid the safe engineering design of reactors. These are used to generate the required estimate of probabilities and source terms of potential severe accidents, in the absence of relevant actual data.
To date, the ExternE Project has made preliminary assessments of major accidents for the nuclear, oil and gas fuel cycles. For the nuclear fuel cycle (European Commission, 1995a), the assessment has used a probabilistic approach. For the oil and gas fuel cycles, the assessment has been based purely on the available accident statistics (European Commission, 1995b). The limitations of the latter approach are recognised and further development of the methodology for assessing accidents is envisaged in the future. The details of the approaches adopted are given in the following sections.

15.2 Accidents in the Nuclear Fuel Cycle

The assessment of potential major accidents has been extensively developed within the nuclear industry. Numerous models exist which assess the consequences of possible releases (for example, COSYMA, CONDOR, MACCS). These models include a probabilistic approach to weather conditions, which can significantly alter the size of potential human health impacts. In addition, possible human actions taken to mitigate the possible impacts (i.e. radiation protection counter-measures) are also included within the models.

When assessing the consequences of severe nuclear accidents, the exact dose-response functions chosen can have a large impact on the overall damages, particularly with long-term effects, that may not be physically discernible from background mortality and morbidity rates. Indeed, large discrepancies have been found between assessments of potential nuclear accidents. This is largely due to the variation in the scenarios and assumptions chosen. Of these, the most important is whether the evaluation should employ a 'worst case' scenario or a more realistic situations.

Risk perception studies have shown that major accidents which result in a large number of deaths are perceived to be worse than many smaller incidents with an equivalent overall death rate. This factor should be considered when assessing the social costs of severe accidents. Some critics argue that the risk-based approach is incomplete, as people often attach a disproportionate importance to events with high impacts, even though the probabilities of such events are extremely low. Such public perceptions are pronounced in the nuclear field, due to a negative perception of nuclear technology.

Such differences between the public perception and expert understanding of potential nuclear risk have been well documented (European Commission, 1995a). Unfortunately little data exist to be able to quantify this difference. The area is currently the subject of considerable debate, as to whether the assessment of risk should include these types of factors. It is often argued that the assessment of risk should be strictly on a technical basis, with expert judgement, so that informed decision-making can be encouraged. Other factors should not be ignored but they should not be directly taken into account in the decision making or political processes. Conversely, others argue that factors reflecting public perception should be included.

An initial proposal for a model which includes such social factors in the estimation of the external costs of severe accidents was made by Krupnick et al (1993). Although the data does
not presently exist to apply and test this model, it is an important step forward in incorporating social factors into evaluations.

A comprehensive Probabilistic Safety Assessment (PSA) of potential reactor accidents is beyond the scope of this project. In addition, the detailed data on potential source terms and associated probabilities for a multitude of potential scenarios for nuclear power plants are not available. However, to provide an indicative estimate of the potential impacts from a severe nuclear accident, probabilities from PSA studies have been used. The COSYMA accident consequence code has been applied for general European conditions, and has been used to estimate the public health impacts and economic consequences of the release. Full details of this assessment are described in the nuclear fuel cycle report (European Commission, 1995a).

15.3 Major Accidents in the Oil and Gas Fuel Cycles

Three categories of major accidents have been identified within the oil and gas fuel cycles:

- Major accidents from offshore production;
- Major accidents from offshore transportation;
- Major accidents onshore.

The offshore activities dominate the major risk analysis for the European oil and gas fuel cycles, though other fuel cycle stages are important. Such potential impacts are large; for instance, the worst accident affecting the UK energy industry within the last thirty years was the Piper Alpha rig explosion.

Preliminary assessment of such major accidents has been undertaken in the ExternE Project based on available accident statistics. Full details are given in the oil and gas fuel cycle report (European Commission, 1995b). The types of accidents which may occur and the methodology which has been used in preliminary assessments are outlined in the following sections.

15.3.1 Major accidents from offshore production

The following events are likely to be the most important in the initiation of accidents on offshore platforms:

- The release of hydrocarbons from process equipment and pipelines, including blow-outs;
- The failure of structures or utilities (including from faults in the construction);
- Helicopter accidents (transportation);
- Collision by ships.

Ideally, a safety assessment of such major accidents should be site specific and include analysis of rig design. Risk is typically assessed by the offshore industry using formalised quantitative risk assessment (QRA), which looks at the probabilities of individual parts of the system failing, and the consequences of such failure. These factors are then combined to give an overall assessment of risk. In order to determine the risk, it is necessary to determine failure frequencies from the limited data bases that exist for accident statistics and plant
reliability. In practice, the techniques also consider hazard and operation (HAZOP) studies, fault tree analysis, human factors and safety audits. The implementation of QRA is beyond the scope of this study, though QRA’s are normally required for any reference platform. It is hoped that data from such studies can be incorporated in future assessments.

To date, the study has assessed major accidents using statistics to provide a simple derivation of probability from historical events. Such an approach should be based upon statistics relevant to a specific field, its location, and relevant technology and safety implementation. However, insufficient data is available to allow this. Disasters of the magnitude of the Piper Alpha disaster are of very low probability and there is therefore little historical information. For this reason, averaging over the 5 year time span recommended for general occupational accidents is unrepresentative. Instead, fatalities from major accidents between 1970 and 1990 are linked to offshore production during the same period to present a normalised risk estimate.

It is interesting to note that most of the major world-wide accidents in the offshore oil and gas industry during the last 20 years have occurred in the Norwegian and UK sectors of the North Sea. This higher incidence is due, largely, to the extremely harsh operating environment of the area. Thus, for the analysis undertaken in the ExternE Project, major accidents have been linked to production within the North Sea to present a more location specific analysis.

For the basis of this analysis, a major accident has been defined as an event involving more than 5 fatalities. This is lower than the definition of 10 fatalities presented earlier but in this case provides a sensible cut-off point for the separation of routine operation and major events. The major North Sea accidents over the past twenty years are shown in Table 15.1.

Table 15.1 Major accidents in the North Sea associated with the extraction of oil and gas between 1970 and 1990.

<table>
<thead>
<tr>
<th>Name</th>
<th>Event</th>
<th>Date</th>
<th>Fatalities</th>
</tr>
</thead>
<tbody>
<tr>
<td>Piper Alpha (UK)</td>
<td>Explosion and fire</td>
<td>1988</td>
<td>167</td>
</tr>
<tr>
<td>Alexander Kielland (Norway)</td>
<td>Structural failure and capsize</td>
<td>1980</td>
<td>123</td>
</tr>
<tr>
<td>Chinook helicopter (UK)</td>
<td>Crashed into sea</td>
<td>1986</td>
<td>45</td>
</tr>
<tr>
<td>Wessex 60 helicopter (UK)</td>
<td>Crashed into sea</td>
<td>1981</td>
<td>13</td>
</tr>
<tr>
<td>CONCEM (Norway)</td>
<td>-</td>
<td>1985</td>
<td>10</td>
</tr>
<tr>
<td>Helicopter (UK)</td>
<td>Crashed into platform</td>
<td>1990</td>
<td>6</td>
</tr>
<tr>
<td>Deep Sea Driller (Norway)</td>
<td>Overturned</td>
<td>1976</td>
<td>6</td>
</tr>
<tr>
<td>Byford Dolphin</td>
<td>-</td>
<td>1983</td>
<td>5</td>
</tr>
<tr>
<td>Statfjord, 33/9A, A (Norway)</td>
<td>Fire</td>
<td>1978</td>
<td>5</td>
</tr>
</tbody>
</table>

TOTAL 380


The total of 380 fatalities in the period 1970 - 1990 is larger than many similar estimates in the literature. The reason for this is the inclusion here of helicopter crashes. These are generally not included in offshore accident statistics if the incident away from the immediate vicinity of the platform. However, as these events are associated with the transport of
personnel to and from the workplace, we consider that to exclude them would lead to a large underestimation of risk. Such transport is particularly important in this project, given the especially harsh flying conditions which prevail over the North Sea.

The offshore production figures for the North Sea during this same period have been used to produce average fatalities per unit of production. The values produced represent a long term and total-North Sea average and not a marginal impact from power station operation. They also represent the upper estimate of damages as significant advances have been made in offshore safety over the past twenty years, including recent changes in working practices following the recommendations of the Cullen Report into the Piper Alpha disaster (Taylor, 1991). They are therefore only regarded as a preliminary estimate and highlighted as an area which warrants further investigation in future studies.

All fatalities have been valued using the Value of Statistical Life (2.6 MECU) and no account has been taken of risk aversion in the valuation of fatalities. It is to be noted that no account is taken of injuries that occur during major accidents, though in comparison to the damages arising from a significant number of deaths it is likely that the associated costs would be small.

15.3.2 Major accidents from offshore transportation

Severe accidents on oil tankers are also an important occupational hazard within the oil fuel cycle. Such events mainly arise from fires and collisions. The North Sea contains some of the busiest shipping routes in the world, and numerous major shipping accidents occur each year. The oil fuel cycle report (European Commission, 1995b) presents a list of world-wide historical accidents on oil tankers and normalised accident and fatality rates against oil trade movements. As with the discussion of offshore accidents above, this represents a long term and world-wide average rather than a marginal impact due to the reference power plant operation.

15.3.3 Major accidents onshore

The potential also exists for major accidents during natural gas transportation, treatment and storage. Similarly, events can also occur during oil refining processes. For both fuel cycles, it is the release of hydrocarbons from process equipment and pipelines which is the cause of major accidents. Any accidents involving fire and explosion may lead to a considerable number of fatalities and injuries. Accidents may occur not only in process areas but also in storage facilities. For natural gas, failures of high pressure mains system are very rare; historically most accidents caused by gas leakage have occurred in the low pressure distribution system.

To date, no attempt has been made to quantify these onshore major accidents within the ExternE Project. Certainly on a global scale they occur regularly, and there are numerous examples within the EU of recent major accidents at such installations. For consistency, such events should be considered in future assessments.
15.4 Conclusions

To date, the treatment of major accidents has been limited to examples from the nuclear and oil and gas cycles. For the nuclear cycle, a set of hypothetical nuclear accidents have been considered and results from probabilistic safety assessment studies have been used to provide an indicative estimate of the potential impacts. In the oil and gas cycles a preliminary analysis has been performed using historical accident rates in the offshore industry; however, it is recognised that some quantitative risk analysis should be used to increase the accuracy of future estimates.

In both cases the results of the assessments are considered to be only indicative of the magnitude of the values that might be obtained. Further work on the treatment of major accidents is to be undertaken in the future ExternE Project.
15.5 References


