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1. INTRODUCTION

1.1 Status of this Report

This report updates the earlier ExternE Methodology report (European Commission, 1995a). It provides details of advances in the methodology made during the third phase of the study which ran from 1996 to 1997. The earlier report showed that it was possible to quantify impacts and their associated externalities using a sophisticated approach that accounted for the latest developments in environmental research. However, it was acknowledged that further work was needed to improve the treatment of a series of important issues, including;

- global warming damages
- major accidents
- non-environmental externalities
- effects of tropospheric ozone on health and crops
- sustainability
- valuation of mortality associated with air pollution
- effects of a broader range of pollutants, such as dioxins and heavy metals
- fuel chain impacts outside the European Union
- uncertainty

The progress made in these areas forms a substantial part of the present report. Note that a separate report on assessment of global warming within ExternE (European Commission, 1998a) is being issued simultaneously. For completeness the executive summary from that report is reproduced here.

It was also necessary to ensure that the methodology reported in 1995 was kept up to date with the latest research findings outside of the ExternE project. This was most important in respect of the health effects of the major air pollutants. In 1995 most of the literature in this field originated from the USA. Since then a number of European studies have been completed, most significantly the EC DGXII APHEA project. The project team feel that without this updating, the earlier project outputs would now be of limited value.

1.2 Objectives of the ExternE Project

The ExternE Project set out to be the first systematic approach to the evaluation of external costs of a wide range of different fuel cycles. The study's principal objectives to the end of 1995, when the first series of reports was published, were:

- To develop a unified methodology for quantifying the environmental impacts and social costs associated with production and consumption of energy;
- To use this methodology to evaluate the external costs of incremental use of different fuel cycles in different locations in the European Union;
- To identify critical methodological issues and research requirements.

The emphasis in the objectives was on methodology. A particular concern was 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 next phase of the study (the phase with which this report is chiefly concerned) ran from 1996 to 1997, and was in three parts;

1. The Core programme, which was oriented toward refinement of the methodology, particularly the issues listed in Section 1.1. A second objective was to explore the issues surrounding the application of impact assessment and externalities data. A third was to apply the methodology to parts of the energy sector that had not been explored previously within ExternE (see European Commission, 1998b).
2. The National Implementation programme, in which the project methodology was applied to the power sector in 15 countries (all European Union member states except Luxembourg, but including Norway; see European Commission 1998c).
3. The ExternE transport programme in which the methodology was adapted for characterisation of the impacts and damages of the transport sector (IER and others, 1998).

The remit of this project is broadly defined and includes all burdens imposed by energy related activities 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). It also includes effects of burdens other than pollution, such as effects of accidents in the workplace. 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 externalities.

The ExternE Project study is closely linked with other work being undertaken within the European Commission's research programmes. For example, a specific objective of the project in the period 1996 to 1997 was that results should be integrated with the new generation of energy-economy-environment models (e.g. GEM E-3 and PRIMES) that were under development within the JOULE programme.

1.3 External Costs: Concept and Application

1.3.1 Defining External Effects

The original scope of this Project was to value the *external costs* of fuel cycles. Hence it is important to be clear what is meant by that term and what difficulties arise in interpreting it. An external cost, or an externality (they are treated as equivalent terms in this study), arises when the social or economic activities of one group of persons have an impact on another group *and that impact is not fully accounted for by the first group*. Thus a power station that emits SO_2 causing damage to building materials or human health, is generating an externality because the resulting impacts are not taken into account by the electricity generator when deciding to operate the power station. This immediately raises the question, what if the generator acts to reduce emissions because of the impacts caused? This action may be in response to social pressure, or because of government regulations as to what can be emitted. There are likely still to be some impacts, but is there an externality? The answer from the economics literature is that if regulations or moral pressure are such as to reduce emissions by the optimal amount then there is *no relevant externality*[1] in the sense that the measured damages should be used to enforce a *further* tax on the polluter, or to argue the case for *further* controls on emissions. Although this may seem self-evident at one level, it has the important implication that environmental costs and externalities are not synonymous, and that measuring the former is not equivalent to identifying the latter.

The issues involved can be seen more clearly with the aid of a simple diagram. Figure 1.1 below plots the damages caused by each increase in the level of air pollution against the level itself, damages are measured in money terms. For most conventional cases (see Tietenberg, 1992, for details), it is generally assumed that additional (or marginal) damages increase with pollution (at least up to a point). The argument is partly taken from a parallel with other goods that affect individual welfare. As the amount of such goods increases, so the value of an additional unit declines. For emissions of pollution, a *decline* is equivalent to an increase in environmental quality. So, as emissions decline, it is argued, the value of additional units of decline should fall, giving an upward sloping shape for the marginal damage curve. However, for several impacts that have been analyzed in this Project, this argument does not hold. In these cases physical damages are a linear function of concentration. This applies to health effects of radionuclides, particulates and several other air pollutants. Concentrations are linearly related to emissions for most dispersion processes. The exceptions are complex phenomena that involve non-linear effects, in particular ozone formation. Since linearity also holds in practice for most of the economic valuations, the marginal damage curve is most likely to be horizontal (Figure 1.1[2]).

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[1] The term relevant externality is a technical term in the economics literature and has been used for that reason. However, it does not imply that the measurement of the external effect is not useful or important even when the regulation has controlled pollution to the optimal amount.

[2] The most typical case of an important non-linearity is where there is a threshold level in the total damage function, implying zero damages below the threshold level and damages increasing (possibly linearly) above it. In such a case the marginal damage function would have zero damages below the threshold and a positive and constant one above it.

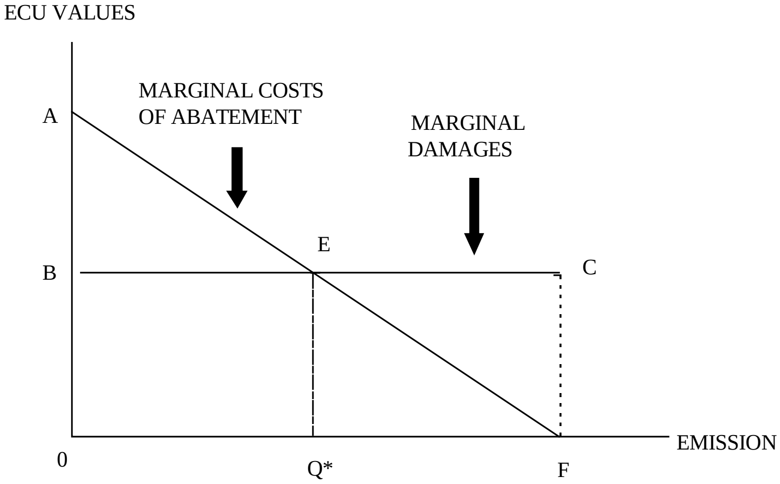

Figure 1.1 Environmental Damage and Costs

On the same figure are plotted the additional or marginal costs of reducing pollution by a small amount (the marginal costs of abatement). Starting from any level of emissions, a cost efficient strategy would be to undertake the least cost reduction options first, and bring in more costly options as the amount of reduction required increased. That would imply a marginal cost of abatement curve that was sloping as shown, and in fact empirical evidence supports that shape. If there are no controls, and if the polluter feels no social obligation to reduce his emissions, the observed level will be OF. At that point the costs of a reduction in emissions to OQ^* are given by the area FEQ^*. At the same time, however, the benefits of the reduction in terms of reduced damages are given by $ECFQ^*$. Hence a level of emission equal to OF represents a position from which a reduction is justified. However, a level of emissions equal to OQ^* does not because at that level there is no decrease (or increase) in emissions that will generate an increase in benefits that are greater than the costs. In other words, OQ^* represents the optimal level of emissions. If the polluter is already at that level, the measured external cost should not be imposed on top of all the other regulations. However, measuring the external cost is still useful in determining what the optimal charge or tax on the polluter should be (see below).

What is the implication of this discussion for the quantification of the external costs of using energy? The purpose of the analysis is to measure the environmental damages created by the emissions at the point at which the plant is operating. In terms of Figure 1.1, one is measuring the area under the damage curve, or OBCF[3], *if the polluter is operating with no controls*.

[3] In fact the measurement has a time dimension and will consist of several pollutants, but the principle remains the one shown in Figure 1.1.

With a horizontal marginal damage curve, the average damages will also be equal to the marginal damages (area OBCF divided by OF) and this will give the correct tax that should be imposed on the polluter (i.e. OB or Q^*E). The optimal tax should be equal to the marginal damage Q^*E because that would result in a reduction in emissions to OQ^*. Each polluter would find that it pays to make every unit of reduction in emission to the right of Q^*, but not to make the reductions to the left of Q^* (the costs of each unit reduced exceed the tax saved). With the assumptions of linearity in the marginal damage curve, the same result will follow even if the polluter is operating at the optimal point Q^*. In such a case, a tax of Q^*E would, of course elicit no direct change in emissions. The polluter would simply pay the tax. The tax itself could, however, have an impact on the profitability of the enterprise and therefore on emissions. This is something that would have to be analyzed.

It is important to recognize that the above analysis depends strongly on the constancy of the marginal damage curve. If the marginal damage is not constant, or if there are thresholds, the estimated marginal damages will depend on the level of emissions and relevance of the calculated value for tax or regulatory policy will be less clear.

In summary therefore, it appears that the estimation of external costs is of interest to policy makers, *whether the polluter is in a regulated state or not*. The *implications* of any estimated values for further regulation, or for a justification of existing regulations will however, require further analysis, which is beyond the scope of this study. In particular it will be important to know what the marginal damage curve looks like and whether a constant marginal damage level is a reasonable assumption.

In this study the terminology used for any estimated damage is that it is an external effect but not necessarily one that implies the need for further controls.

1.3.2 Internal Versus External Effects

A related issue that arises in going from environmental damages to external effects is that of internalization. Damages may occur to certain groups but the latter may have fully taken account of the damages in deciding on what actions to take. A central case concerns workers in certain industries who suffer from higher incidence of diseases, or higher frequency of accidents. If they are aware of these factors, and if the contract of employment reflects proper compensations for these added risks, then there is no relevant externality in the sense defined above. In technical terms it is said that the externality has been internalized through a contract that has been entered into freely. It may still be of interest to measure the ‘social costs’ for such impacts, as it was for the externalities where the polluter is operating under regulation, but care has to be taken in what interpretation is placed on the resulting damage figures.

Two matters of contention in deciding whether or not an externality has been internalized or not, are those of information and market imperfections. It is argued that if a worker takes a job not knowing the risks involved, and then suffers from work related health effects, there is a genuine external effect that needs to be accounted for. It is unarguable, for example, that the presence of an unexpected impact on a working population which was not foreseen is an external effect. What is less clear is the significance of such effects for policy. Clearly, if neither party is aware of a possible effect, and the same applies to the regulating authority, then no measurable external damages can be included in the analysis. If the regulating authority is aware of certain health risks, and believes the working population is not, the appropriate measure is to disseminate the information. The resources that are devoted to that, however, could be influenced by the size of the environmental damages caused to this group. In this sense then, data on work related 'externalities' may be relevant to the analysis.

A related issue is that of market imperfections. Staying with the example of occupational health effects, there may be macroeconomic difficulties in the region, in which case those seeking work would not have the luxury of choosing between a riskier occupation with a higher wage, and a less risky one with a lower wage. This is also pertinent to deciding whether or not an external effect is present. In the presence of structural or long term unemployment, as is the case in some parts of Europe, the occupational health damages will form part of the external costs of the related fuel cycles. In other situations, however, where the labor force is more mobile, and where unemployment is less long term, the environmental damages may be said to be internalized.

1.3.3 Application of External Costs Data and Methodologies

The above discussion is used here purely for the purposes of illustration, rather than to define boundaries to the use of the data generated within the ExternE Project. Although the study commenced from the need to provide environmental elements to E3 (energy, environment, economy) models, it soon became apparent that the information generated within the study was useful elsewhere. Indeed, since 1995 the study has found numerous applications, including;

- Economic evaluation of air quality limits for SO_2, NO_2, particles and lead
- Cost-benefit analysis of a draft directive for governing emissions from waste incinerators
- Assessment of electricity generating options in remote communities, such as Crete
- Consideration of funding arrangements for renewable energy technologies

1.4 Research on Externalities

1.4.1 Previous Studies

Growing world-wide concern about the environment led to several studies in the late 1980s and early 1990s which have estimated some of the externalities associated with electric power production and fuel cycles. The most prominent of these were by Hohmeyer (1988), Ottinger *et al* at Pace University (1990), Bernow *et al* (1990) of the Tellus Institute, ECO Northwest (1987) and Pearce *et al* (1992).

Hohmeyer's study used a 'top-down' approach. The key steps of his analysis of fossil power plants were as follows:

- Development of an inventory of emissions of carbon monoxide (CO), particulate matter (PM), oxides of nitrogen (NO_x), sulphur dioxide (SO_2), 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 1.2. 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.

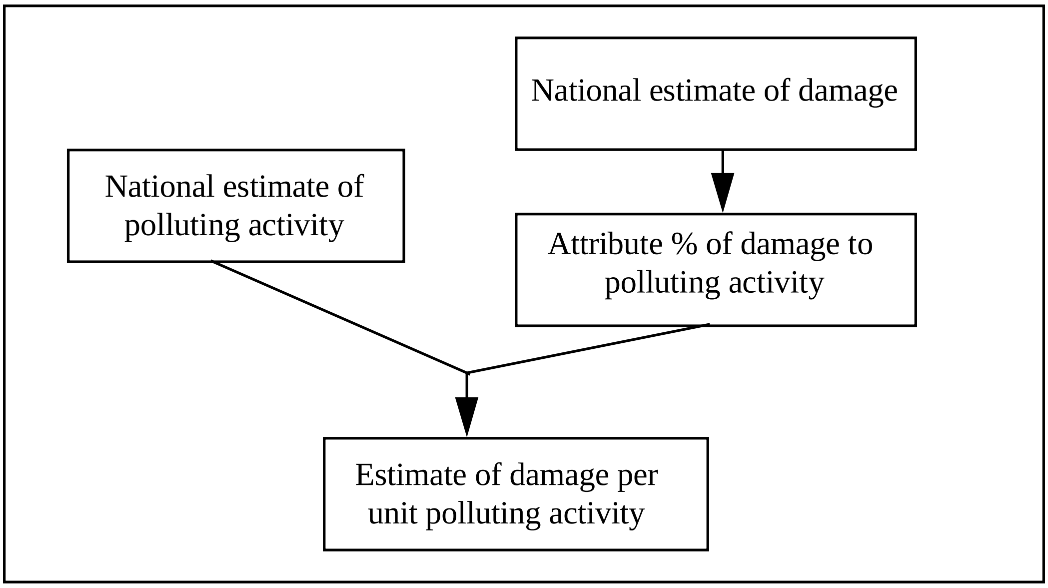

Figure 1.2 Illustration of the 'top-down' approach for externality assessment. This approach takes no account of the site specificity of many types of impact.

Since the analysis was done in terms of national averages, effects due to variations in population density and pollutant concentration were neglected. The transport of pollutants across national boundaries was also neglected.

Pace relied entirely on numerical estimates from previous studies (e.g. ECO Northwest, 1987) and used those values to compute damages. The study by the Bonneville Power Administration (1991) also relied on ECO Northwest's studies.

The study by Pearce *et al* (1992) was similar in spirit to Pace and drew quite extensively on it. However, by taking a fuel cycle approach, they addressed more impacts. In neither case were data collected at the primary level. Thus these studies are not site specific and do not take account of differences in external costs that arise from differences in topography or concentrations of population.

Bernow *et al* (1990) of the Tellus Institute pointed out that it is difficult to estimate social costs based on damages. They suggested that abatement costs may be a reasonable surrogate for damages. In this approach, existing and proposed environmental regulations are analysed to estimate the value that society implicitly places on different environmental impacts. According to Tellus, the marginal cost of abating emissions, when they are at the limit imposed by regulation, reflects the preference of regulators to require that particular level of abatement and the corresponding incremental cost, rather than allow emissions to exceed that limit and subsequently to have adverse impact on the public. The reasoning used by Tellus is that since these regulators represent the public, their views represent the costs placed on those emissions by the public.

There is now a considerable consensus that such reasoning is flawed. The premise that marginal control costs represent the costs of air emissions to society implies that regulators know what individual environmental damages are and always decide on the optimal policy where the marginal costs of control equal the marginal damages. In fact it is quite clear that they do not know these costs, and the political processes by which policy decisions are made do not generally have the property that they equate social damages to costs of abatement. 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.

1.4.2 Recent Research on External Costs

It follows from the above that *basic research* in this area was needed to develop a methodology that made much better use of the available science. From its beginnings, the ExternE study went beyond the earlier studies in several respects. These are:

a) *A more thorough characterisation of the energy technologies and their discharges into the environment on a site specific basis*. Clearly the environmental impacts of electric power generation will vary according to the technology adopted. These need to be made much more precise than has been done in the past before valuations can be carried out. Furthermore, the impacts will vary according to where the plant is located. On *a priori* grounds, site specific differences should be important. This study aims to find out how important these differences are by evaluating the damages from plants with the same technology but with different locations.

b) *A consideration of all major stages of a fuel cycle rather than just electric power generation.* Significant environmental impacts occur during mining of fuels, their transportation and the eventual disposal of wastes. These need to be evaluated as carefully as those of the generating stage. In some cases, particularly for the renewable fuel chains, the generating stage may cause little environmental impact.
c) *Modelling the dispersion and transformation of pollutants over their full range.* Earlier work relied mainly on short distance dispersion models for the majority of the air pollution impacts. However, these were inadequate, as reference to the concern over trans-boundary air pollution in Europe clearly shows. Chemical reaction of pollutants in the atmosphere also needs to be accounted for.
d) *Engaging in a more extensive, critical review and use of the ecology, health sciences, and economics literature than previous studies*. Many ecological, epidemiological and valuation studies have come out in the last ten years which needed integration within a framework whereby they could be better used for policy analysis.
e) *Estimating externalities by accounting for existing market, regulatory, insurance and other conditions that internalise some damages so that they are not externalities*. The issue of what is and what is not an external effect has been discussed earlier in this chapter. It arises not just in the occupational health impacts case that was cited but also in connection with special provisions made to compensate victims of accidents caused during fuel cycle operations. This study makes an attempt to analyse when, and to what extent, such provisions may have internalised the external effect.

Since 1991 a number of other studies have been undertaken using a broadly similar approach to ExternE. The main examples are the companion US DOE study (Oak Ridge National Laboratory and Resources for the Future, 1992, 1994a, b, c and 1995) and the New York State Study (Rowe *et al*, 1995). The most prominent example outside Europe and the USA is probably the study by van Horen (1996) in South Africa.

1.5 Purpose and Structure of the Report

The objective of the present report is to provide details of the methodology developed and used by the ExternE Project. 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, 1995c-f, 1998a-c). As stated above, further information on the assessment of global warming damages is given in European Commission (1998a).

This report also provides an opportunity to clarify some important misunderstandings that arose in connection with the earlier series of reports. Examples include:

- "*Health assessment was based on review of only a small number of studies*."
  The health assessment was in fact based on a thorough review of the available literature by experts in both Europe and the USA. The single studies that were cited in the tables of recommended functions were selected as being representative of the wider literature.

- "*ExternE did not consider global warming damages*."
  ExternE 1995 considered, quantified, and reported global warming damages, though results were derived from earlier studies (i.e. unlike now, the estimates given were not original to the Project). Then, as now, wide ranges were reported for global warming, reflecting the state of scientific assessment in this field.

- "*No account was taken of fuel cycle stages outside the country where the generation facility was based*."
  This misunderstanding arose because of the cases considered in the earlier fuel cycle assessment reports (European Commission, 1995c-f). At the time of writing the first series of reports the assumption that all stages of the fuel cycle were likely to be in the same country as the power generation unit was realistic for the UK and German coal cycles, the French nuclear cycle and so on. Note that this assumption was not adopted for the German assessment of the oil fuel cycle (European Commission 1995d). The recommendation then as now was actually that the analysis should consider realistic locations for each stage, no matter where in the world they take place.

- "*The ExternE Project found that the external costs of the fuel cycle are X mECU/kWh*".
  The Project does not recommend single estimates of fuel cycle damages for purposes other than illustration of trends. For application by policy makers it is essential that ranges are used and sensitivities explored, because of the extensive uncertainties that affect external costs analysis.

The following Table summarises the structure of the report and identifies the changes that have been made compared to European Commission (1995b). A major difference in structure is that the economic valuation section in the earlier report has now been fully integrated with the impact assessment report.

Table 1.1. Structure of this report, with summary of changes made since European Commission (1995b).

Chapter	Changes
Executive Summary	Fully revised
1. Introduction	Fully revised
2. The Impact Pathway Methodology	Some issues clarified, commentary on sustainability added
3. General Issues of Economic Valuation in Assessment of Fuel Cycle Externalities	Integrated from earlier separate section on economic valuation
4. Models for Air Pollution Analysis	Review of EcoSense model now included
5. Assessment of Uncertainty	New chapter
6. Assessment of Global Warming Damages	Fully revised
7. Major Accidents	Fully revised
8. Health effects of PM_{10}, SO_2, NO_x, O_3 and CO	Revised to include more pollutants and new epidemiological data, particularly from Europe
9. Health Effects of Heavy Metals, Dioxins, and Other Atmospheric Micropollutants	New chapter
10. Radiological Health Effects	As European Commission (1995b)
11. Occupational Health Effects	Minor revisions only
12. Economic Valuation of Health Effects	Major revision
13. Effects on Terrestrial Ecosystems	Major revision
14. Effects of Water Use and Pollution	More complete integration of rationales and methods for dealing with aquatic impacts
15. Effects of Air Pollution on Building Materials	Updated
16. Quantification of Ozone Damages	New chapter
17. Noise and Amenity	As European Commission (1995b)
18. Visual Amenity	Visibility effects now included
19. Non-Environmental Externalities	New chapter
20. Aggregation of Externalities Data	New chapter
21. Conclusions	New chapter

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2. THE IMPACT PATHWAY METHODOLOGY

2.1 Approaches Used for Externality Analysis

As discussed in Chapter 1, 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.

Life cycle analysis (LCA - OECD, 1992; Heijungs *et al*, 1992; Lindfors *et al*, 1995) provided a starting point for development of the overall ExternE Project methodology. LCA 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.

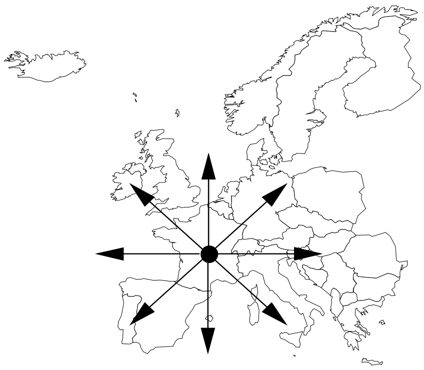
One of the major issues in any LCA concerns the way in which the ‘system boundaries’ are specified, defining the extent of the analysis around the basic system under investigation. Some limit on the system is clearly essential, otherwise analysis becomes impracticable. However, much thought needs to 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. Results are often highly specific to the system boundary that has been defined and so great care is needed in the setting of boundaries.

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:

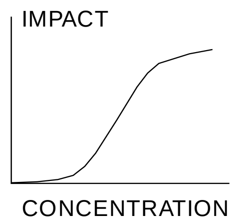
1. Definition of objectives and system boundaries;
2. Collation of data to provide an emissions inventory;
3. Classification of emissions into impact categories;
4. Characterisation of impacts;
5. Normalisation;
6. Valuation (translation of impact assessments into a common unit).

EMISSIONS
(e.g. tonnes/year of SO_2)

DISPERSION
INCREASE IN AMBIENT CONCENTRATIONS
(e.g. ppb SO_2 for all affected regions)



IMPACT
(e.g. change in crop yield)



COST

Stage 4 of LCA, '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, NO_x, SO_2 and ozone (O_3). 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 met

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 (European Commission, 1995a - d) 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 ExternE has been 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.

In our analysis we have therefore set out to identify realistic sites for each fuel cycle activity. 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. 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 (UN ECE) Convention on Long Range Transboundary Air Pollution, demonstrate the importance attached to site dependence by decision makers.

2.3.3 Technologies

The methods defined here are applicable to any technology, old, new, or even of the future. Note also that the methodology has now been applied simultaneously to many emission sources (Chapter 20), providing aggregate estimates of damages, rather than the single plant estimates previously calculated. This new capability much strengthens the study as a tool for advising policy makers.

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, availability of fissile material for non-peaceful purposes).

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 (indeed there is a very strong rationale for regarding this 'burden' as trivial). 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. That this may well complicate the analysis is not an issue: the purpose of the assessment must be to obtain as reliable an impression of externalities as possible, and if this means requiring data on occupational health effects in a far distant country then so be it. Note that if, in such a case, country-specific data are not obtainable, it is usually best to extrapolate from whatever data are available (recording that this has been done and hence that the results are subject to perhaps significant error).

The importance of considering trans-national impacts was illustrated in our first series of reports by the demonstration that some air pollutants can cause significant damages even at great distances from the point of emission. This had of course been accepted for many years with respect to ecological damage in Northern Europe, though it had not previously been translated into externalities. 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

2.4.1 Prioritisation of Impacts

It is possible to produce a list of several hundred sources of burdens 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. At this point there comes a divergence between the ExternE Project methodology and LCA: this simply reflects the contrasting objectives of the two methods. Wherever possible scoping calculations should be made or referenced to gain some idea of the likely magnitude of impacts that have been omitted from subsequent analysis.

There is the possibility for serious errors to arise at this point. Reference to existing analyses as a rationale for excluding a given effect as likely to be insignificant may or not be appropriate depending on a range of factors. The legislative framework within which facilities operate is likely to be particularly important as this varies substantially around the world. An obvious example concerns occupational fatality rates which may vary by a factor of 100 or more between countries (see Holland *et al*, 1998). Within ExternE we have typically paid little attention to damage arising from liquid discharges, though these may be significant where controls over waste water emissions are less stringent that those that operate in the European Union. It is thus apparent that the prioritisation exercise requires much care it if is not to induce an unsuspected level of error.

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 may 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_2 from a power station as a priority burden, why not include emissions of SO_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 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 2.2. 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 2.3, illustrating the series of processes that need to be accounted for from emission of acidifying pollutants to valuation of impacts on freshwater ecosystems.

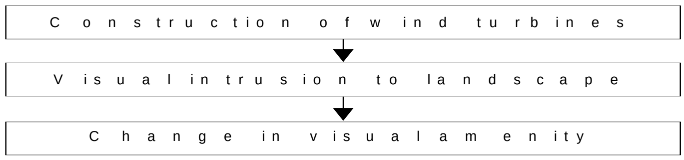


Figure 2.2 The impact pathway describing the processes associated with changes in visual amenity resulting from the development of a wind farm.

A number of points should be made about Figure 2.3. 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, reflecting on the current state of knowledge of the impacts addressed. The analysis can easily be extended once further data become available. For legibility, numerous feedbacks and interactions are not explicitly 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.

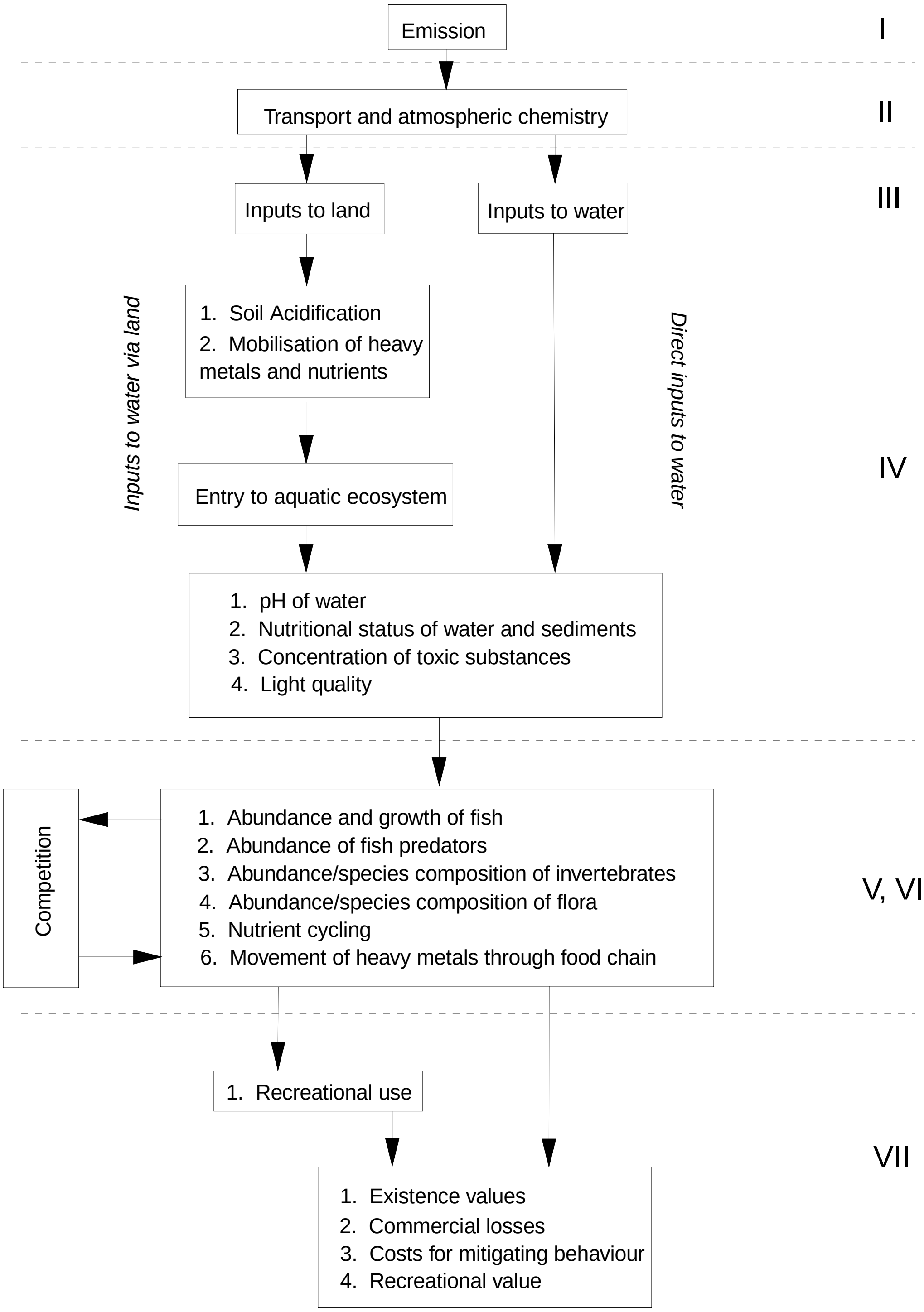


Figure 2.3 The impact pathway showing the series of linkages between emission of acidifying pollutants and the valuation of impacts to freshwater fisheries. This complex example contrasts with the simple scheme shown in Figure 2.2.

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 (CO_2, SO_2, NO_x, CO, volatile organic compounds and particulate matter) are quite well known. Especially well determined is the rate of CO_2 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, though this simply reflects reality and so efforts must be made to quantify with the least errors possible. 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.

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 - 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. These have been included in the EcoSense software (Chapter 4).

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 exceedence 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 2.2 and 2.3, 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 4 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 2.4. 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.

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.

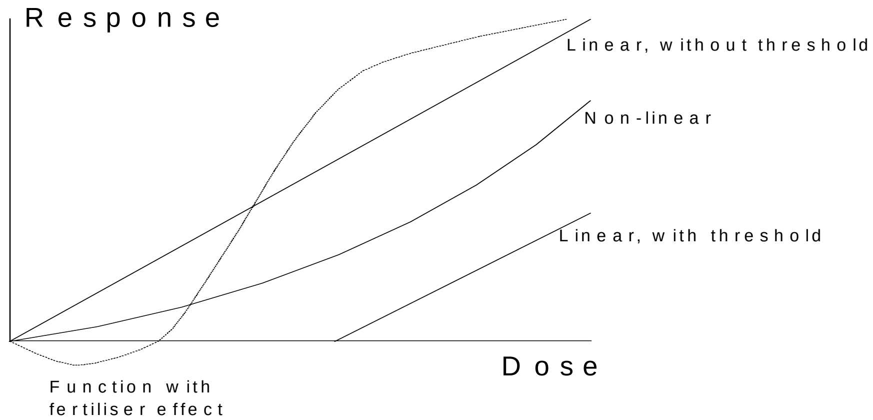


Figure 2.4 A variety of possible forms for dose-response functions.

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 Chapter 3 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.

Further details on the treatment of uncertainty in the study are given in Chapter 5.

2.5 Impacts Assessed in the ExternE Project

This section lists the priorities identified for several fuel cycles for electricity generation. The study has now gone beyond the fuel cycles discussed here, into transport, waste, end use technologies and so on. However, the listing shown is sufficient to demonstrate the diversity of different parts of the energy sector, and the fact that it is not possible to draw up lists of priority impacts that are common to all analysis.

2.5.1 Priority Impacts for Fossil Technologies

The following list of priority impacts was derived for the fossil fuel cycles in our earlier reports (European Commission, 1995 a, d). It is necessary to repeat that this list was compiled for the specific fuel cycles considered at that time, and should be reassessed for any new cases. The first group of impacts is 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;
9. Impacts of noise.

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 (European Commission, 1995c) 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 were 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 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 Consistency

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, and remembering that we are here concerned with externalities rather than life cycle 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 The ExternE Methodology and Sustainability

As a result of the United Nations Conference on Environment and Development in 1992, most governments world-wide have adopted sustainable development as a national goal. These governments are now struggling with the design of policies to achieve this goal and it has become increasingly clear that decision-makers need indicators of sustainability that can guide their actions. A specific task of ExternE in the period 1996 to 1997 was to consider how the study methodology fitted with the thrust of development of sustainability indicators. This section summarises a longer report by Atkinson *et al* (1997).

2.6.1 Sustainable Development Framework

Sustainable development can be defined as *non-declining human well-being over time*, a goal which in turn reflects a concern for future generations. Difficulties arise when one considers how this goal is to be achieved - i.e. what are the conditions for attaining sustainable development - and in unravelling the ethical problems inherent in implementing policies for sustainability. Broadly, two schools of thought exist:

- *weak sustainability:* this states that total (usually interpreted as, national) wealth should not decrease over time. So long as adequate compensation is made in the form of saving and investment it does not matter if say, produced capital or human capital are substituted for natural assets.
- *strong sustainability:* this suggests a greater emphasis on the conservation of natural assets within the broader goal of prudently managing a (nation’s) portfolio of assets over time. Specifically, it is argued that some classes of natural assets have no substitutes and therefore cannot be replaced: i.e. critical natural assets.

What these approaches have in common is an emphasis on what happens to *wealth over time*. Indicators of sustainability, therefore, tend to emphasise either stocks of wealth (e.g. resources, environmental liabilities) or changes in wealth (e.g. depletion, degradation). However, these frameworks provide very different ‘views of the world’ in which we live and thus, they imply different sets of indicators.

2.6.2 Sustainability Indicators

Table 2.1 illustrates that approaches to developing sustainability indicators can be consistent with the broad focus of the existing ExternE methodology on fuel cycles and types of pollutants considered therein. The list of relevant *impact related pollutants* include sulphur dioxide (SO_2), nitrogen oxide (NO_x), acid and nitrogen deposition, carbon dioxide (CO_2), ground level ozone (O_3), particulate matter (PM_{10}) and radioisotopes. In what follows, indicators of *weak* sustainability are presented in terms of *impacts* (e.g. on health, forests, materials etc.). *Strong* sustainability issues are identified according to policy relevant *environmental themes*. The main themes identified include acidification and eutrophication of both terrestrial and freshwater ecosystems, radiation and global warming. Within themes it also clear from the Table that quantification of impacts is wholly relevant to the development of indicators: i.e. the asset damage or loss examined are those related to ecosystems, human health and fisheries.

Table 2.1 Sustainability indicators framework

Sustainable Development Framework	Environmental Themes	Pollutants	Impacts On	Thresholds	Indicators
Strong	Acidification/ Eutrophication	SO_2 NO_x N deposition Acid deposition	Natural, Semi-natural ecosystems Forests Freshwater ecosystems Fisheries	Critical Loads/Levels, Target loads, Gap closure Acid neutralising capacity (ANC)	Exceedence, Weighted exceedence, Monetary value of ANC exceedence
Strong	Nuclear	Radioisotopes	Human Health	Typical background radiation levels	Level exceedence
Strong	Global Warming	CO_2 etc.	Climate	Acceptable temperature changes	Exceedence of acceptable temperature changes
Weak	....	PM_{10} SO_2 O_3 CO_2 etc.	Human Health - Morbidity - Mortality Forests Materials Climate change	Current disamenity, Asset depreciation	Monetary values of disamenity and depreciation

2.6.3 Thresholds for Indicator Development

The construction of a sustainability indicator must be made with respect to *clearly defined thresholds or reference values* against which sustainability can be judged. Hence, exceedance of specified thresholds might be indicative of non-sustainability. However, it should be noted that there is inevitably some uncertainty in making this judgement. For example, thresholds can be defined relative to some scientific criteria reflecting biophysical constraints on say, natural and semi-natural ecosystems, such as critical loads. Yet, exceedance of a threshold *per se* does not necessarily imply significant harmful effects, although it may increase the risk of such impacts.

Objectivity is a desirable attribute in determining thresholds. This points, therefore, to the efficacy of scientific criteria for evaluating reference points. However, alternative values may be considered as benchmarks reflecting say, *social acceptability*. 'Acceptable' thresholds might reflect either that scientific reference values may not be technically feasible or economically cost effective to achieve. While such reference points will lie, in general, above scientific thresholds, it still may be possible to construct an indicator that allows us to judge whether we are *moving in the direction of 'sustainability'* (even if the goal of policy is not actually to reach this - scientifically determined - point). Hence, politically determined thresholds retain some useful characteristics as reference points for sustainability indicators.

2.6.4 Unit of Measurement

The theories of weak and strong sustainability offer divergent guides to what *unit of measurement* should be used in constructing indicators. The natural unit for weak sustainability - which stresses the possibility of trade-offs between produced and natural assets - is *monetary value*. In contrast, given that strong sustainability stresses that decreases in the stock of *certain* (or *critical*) natural assets is unsustainable, there is little need to express this information in monetary units. Some appropriate indication of *physical* decline could provide sufficient information to signal unsustainable development.

As such it is often claimed that physical indicators have an obvious link to the strong sustainability paradigm. Moreover, a recurrent theme in the strong sustainability literature is that monetary values cannot be held with any certainty. That is, there is too much uncertainty say, regarding the link between impacts on receptors and socioeconomic variables to justify couching indicators in precise monetary terms. These concerns encompass both nature of impacts themselves and how individuals will value these impacts. Hence, it has been asserted that physical indicators tend to be more 'objective', which as argued above is a desirable characteristic. However, it should be noted that this appeal is superficial unless underpinned by a satisfactory theory of strong sustainable development (i.e. providing the rationale for interest in the change in physical indicators). That is, it is an appropriately specified reference point or threshold that allows a physical indicator to be judged in strong sustainability terms.

2.6.5 Aggregation

The presentation of any set of indicators, of sustainability or otherwise, inevitably encounters the question of the appropriate level of *aggregation*. Some aggregation is inevitable in that indicators can be defined as aggregates of more elementary data. Indeed, this is what distinguishes an indicator from ‘raw data’: in that, it should have a significance that transcends that of the individual constituent data. Nevertheless, there are limits to the ‘value-added’ to be gained from the aggregation of such information. For example, while money values can simply be added together, the degree of aggregation that is appropriate will depend very much on policy usefulness. While indicators such as *genuine saving* and *green national income* provide useful inferences on (weak) sustainability at the national level, other policy uses may point to a more disaggregated approach, for example, focusing on *contribution of a particular sector to sustainability*. Indicators, therefore, should reflect these needs as well.

Similar comments apply to the aggregation of strong sustainability indicators, which by token of their physical nature are measured in heterogeneous units. In this context there is also a problem of finding a common unit where indicators are measured in different physical units. If a composite measure is to be derived then some alternative non-monetary scheme of weighting must be devised. Although the appeal of obtaining ‘one number’ is self-evident, its actual usefulness and conceptual basis is often more questionable. If aggregation is contemplated it is clear that any such weighting should only be undertaken if there are justifiable scientific or socioeconomic grounds. It is clear that, in the current context, there is little basis for aggregating indicators *across* themes: e.g. to derive an indicator which is a composite index of acidification and climate change indicators.

2.6.6 Consistency With The ExternE Framework

The ExternE methodology is mainly concerned with the evaluation of the social costs of fuel cycles. The main outputs of these efforts has been the monetary indicators of social costs for specific fuel cycles from reference power facilities for a variety of impacts. This approach therefore seems *wholly consistent with the framework used in the construction of weak sustainability indicators*. Within this framework, indicator outputs have been expressed in terms of monetary values, specifically marginal valuations of damages.

In contrast, while strong sustainability is largely concerned with constraints on the use of natural assets at a relatively disaggregated level, it is the *absolute scale of effect* rather than marginal damages that is the focus of attention. This is reflected in the concept of threshold exceedance. In the context of work conducted in this phase of ExterrnE on the aggregation of impacts and damages (see Chapter 20) there are again clear similarities.

2.6.7 Conclusions

Indicators of sustainable development must be underpinned by an adequate theory of sustainability:
- strong sustainability - investigated via policy relevant *environmental themes* (acidification/ eutrophication, nuclear and global warming);
- weak sustainability - investigated via *pollutants* and *impacts*, e.g. the effects of PM_{10} on human health.

Indicators should be defined relative to *clearly defined thresholds* or *reference values*, such that exceedance of these values indicates non-sustainability:
- ideally, such thresholds should be *objectively* determined;
- alternatively, *target* or *politically* determined thresholds might be considered. While such goals are not strong sustainability thresholds as such, some compromise is to be had in that they would still allow a judgement to made as to whether one is *moving in the direction of strong sustainability*.

The natural units of measurement for strong and weak sustainability are respectively, *physical* and *monetary* values. However, weak sustainability does not preclude the use of physical indicators to supplement monetary values. Indeed, where such values are unavailable, physical indicators provide worthwhile information. Also, it may be possible to gain an indication of the degree of movement towards sustainability by estimating the monetary damage of exceeding strong sustainability thresholds.

2.7 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.

Finally, we have reported briefly a review of externalities analysis adopting the impact pathway methodology, in the context of work on sustainability. We conclude that the methodology that we have defined has much to offer in this area, whether one abides by weak or strong definitions of sustainability. With respect to the latter the attention given in this phase of the work to aggregation has been a major advance.

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3. GENERAL ISSUES OF ECONOMIC VALUATION IN ASSESSMENT OF FUEL CYCLE EXTERNALITIES

3.1 Techniques for Valuation

The underlying principle in monetary valuation is to obtain the *willingness to pay* (WTP) of the affected individual to avoid a negative impact, or the *willingness to accept* (WTA) payment as compensation if a negative impact takes place. The rationale is that values should be based on individual preferences, which are translated into money terms through individual WTP and WTA.

Once impacts have been identified in physical terms, they can be valued using market prices, for example, crop damages can be measured by examining reductions in market prices, or lost income through decreasing yields, though even in this simple case there are problems and issues that arise. For a wide range of impacts, however, such as increased risk of death or loss of recreational values, there are no direct market prices that can be used. Three techniques are widely used in this context. One is to elicit the WTP or WTA by direct questionnaire. This is termed the *contingent valuation method* and has been developed into a sophisticated procedure for valuing a number of environmental impacts. Another technique is to look at the WTP as expressed in related markets. As illustration, environmental effects are often reflected in property values. Thus an increase in noise or a reduction in visibility will 'show up' in reductions in the value of properties affected by the changes. This approach is called the *hedonic price method* and is widely used for noise and aesthetic effects.

Where individuals undertake expenditures to benefit from a facility such as a park or a fishing area one can determine their WTP through the expenditures on the recreational activity, including costs of travel to the park, entrance fees paid etc. Economists have developed quite sophisticated procedures for estimating the values of changes in environmental facilities using such data. This method is known as the *travel cost method* and is particularly useful for valuing recreational impacts.

3.2 Categories of Value

The WTP/WTA numbers can be expressed for a number of categories of value. The most important distinction is between values arising from the use of the environment by the individual and values that arise even when there is no identifiable use made of that environment. These are called use values and non-use values respectively. Non-use values are also sometimes referred to as existence values.

Within the category 'use values' there are many different categories. Direct use values arise when an individual makes use of the environment (e.g. by breathing the air, or by admiring a view) and derives a loss of welfare if that environment is polluted. Indirect use values arise when an individual's welfare is affected by what happens to another individual. For example, if I feel a loss of welfare as a result of the death or illness of a friend or relation, resulting from increased levels of air pollution, then this loss of welfare translates into a cost through my WTP. It can and has been measured in limited cases and is referred to as an altruistic value. Both direct and indirect use values have a time dimension; an environmental change today can result in such values both now and in the future.

Another category of use value that is potentially important is that of option value. This arises when an action taken now can result in a change in the supply or availability of some environmental good in the future. For example, flooding a region to impound water for a hydro project would result in that area not being available for hiking. A person might have a WTP for the option to use that hiking area, even if they were not sure that it would be used. This WTP is the sum of the expected gain in welfare from the use of the area, plus a certain gain in welfare from the knowledge that he or she *could* use it even if it is not actually used. The latter is referred to as the option value. The literature on environmental valuation shows that, in certain cases the option value will be positive but in general it is not an important category of value (Freeman, 1989). There are very few estimates of such values, and in the context of the fuel cycle study it was felt that estimating future use values was difficult enough; estimating option values was not considered an important category to address.

The last category of value is non-use value. This is a controversial category, although values deriving from the existence of pristine environments are real enough, even for those who never make any use of them. In some respects what constitutes 'use' and what constitutes 'non-use' is not clear. If someone sees a programme about a wilderness area but never visits it, that represents a use value, however indirect. Pure non-use value must not involve any welfare from any sensory experience related to the item being valued. In fact some environmentalists argue that such non-use or existence values are unrelated to human appreciation or otherwise of the environment, but are embedded in, or intrinsic to, the things being valued. For a sympathetic review of this position, see the discussion on the 'new naturalist ethics' by Turner and Brooke (1988). However, that is not the position taken in this study. The basis of valuation remains therefore an anthropocentric one which, however many economists argue, does not imply an anti-environment stance.

The difficulty in defining non-use values extends to measuring them. The only method available for this category is that of the questionnaire approach, or contingent valuation. This method has been tested and improved extensively in the past 20 years, and the general consensus is that the technique works effectively, where 'market conditions' of exchange can be simulated effectively and where the respondent has considerable familiarity with the item being valued. For most categories of non-use value this is simply not the case. Hence, for the present, non-use values are extremely difficult to value with any accuracy. A comprehensive review of these issues by Arrow et al (1993) comes to similar conclusions.

3.3 Issues Arising in the Use of Monetary Values

The basic philosophy underlying valuation is based on individual preferences, which are expressed through a willingness to pay (WTP) for something that improves individual welfare, and willingness to accept payment (WTA) for something that reduces individual welfare. The total value of environmental impacts is taken as the sum of the WTP or WTA of individuals. No special weight is given to any particular group. This approach contrasts, for example, with that of values based on expert opinion, or values based on the costs of making good any damage done to the environment by the fuel cycle. Such mitigation costs will only provide a valid measure of cost if society is collectively willing to pay for the mitigation, rather than suffer the damage. In such cases mitigation based estimates can provide important values, and have in fact been used in the study in selected areas. However, the validity of that use is dependent on the assumption that society is willing to pay for the mitigation.

Although the valuation of environmental impacts using money values is widespread and growing, there are still many people who find the idea strange at best, and distasteful and unacceptable at worst. Given the central role being played by monetary valuation in this exercise, a justification of the method is warranted.

One objection often voiced in the use of WTP is that it is ‘income constrained’. Since you cannot pay what you do not have, a poorer person’s WTP is less than that of a richer person, other things being equal. This occurs most forcefully in connection with the valuation of mortality where the WTP to avoid an increase in the risk of death is measured. In general one would expect the ‘value’ for a poor person to be less than that of a rich person. But this is no more or less objectionable than saying that a rich person can and does spend more on health protection than a poor person; or that individuals of higher social status and wealth live longer on average than person of lower status; or that better neighbourhoods will spend more on environmental protection than poorer neighbourhoods. The basic inequalities in society result in different values being put on the environment by different people. One may object to these inequalities, and make a strong case to change them, but as long as they are there, one has to accept the consequences. One could argue, for example, that increased expenditure on high technology medicine in Europe is unethical, even though the citizens of that region have a WTP that justifies such expenditures, because the same expenditure on preventative medicine in a poor developing country would save more lives. However, society does not accept such an argument, taking the view that most decisions about allocation of resources are predicated on the existing inequality of income and wealth, both between and within societies.

3.4 Transferability of Benefit Estimates

3.4.1 Introduction

The environmental damages of a particular fuel cycle will depend on which type of plant is being considered and when that plant is being operated. Clearly, however, it would not be feasible to estimate all environmental damages for each location and time specific fuel cycle *ab initio*.

Much of the work required is extremely time consuming and expensive, making the transfer of estimates from one study to another an important part of the exercise. The difficult issue is to know when a damage estimate is transferable and what modifications, if any, need to be made before it can be used in its new context.

3.4.2 Benefit Transfer

Benefit transfer is 'an application of monetary values from a particular valuation study to an alternative or secondary policy decision setting, often in another geographic area than the one where the original study was performed' (Navrud, 1994). There are three main biases inherent in transferring benefits to other areas:

a) original data sets vary from those in the place of application, and the problems inherent in non-market valuation methods are magnified if transferring to another area;
b) monetary estimates are often stated in units other than the impacts. For example, in the case of damage by acidic deposition to freshwater fisheries, dose response functions may estimate mortality (reduced fish populations) while benefit estimates are based on behavioural changes (reduced angling days). The linkage between these two units must be established to enable damage estimation;
c) studies most often estimate benefits in average, non-marginal terms and do not use methods designed to be transferable in terms of site, region and population characteristics.

Benefit transfer application can be based on: (a) expert opinion, or (b) meta analysis. Expert opinion looks at how reasonable making the transfer is and in determining what modifications or proxies are needed to make the transfer more accurate. In many cases expert opinion has been resorted to in making the benefit transfer during the ExternE Project. In general the more 'conditional' the original data estimates (e.g. damages per person, per unit of dispersed pollution, for a given age distribution) the better the benefit transfer will be. In one particular case (that of recreational benefits) an attempt was made to check on the accuracy of a benefit transfer by comparing the transferred damage estimate with that obtained by a direct study of the costs. The finding there was not encouraging in that the two figures varied by a wide margin.

3.4.3 Meta-Analysis

Where several studies, reporting a similar final estimate of environmental damage, exist, and where there are significant differences between them in terms of the background variables, a procedure known as *meta-analysis* has been developed to transfer the results from one study across to other applications. What such an analysis does is to take the estimated damages from a range of studies of, for example, coal fired plants and see how they vary systematically, according to affected population, building areas, crops, level of income of the population, etc. The analysis is carried out using econometric techniques, which yield estimates of the responsiveness of damages to the various factors that render them more transferable across situations. This can then be used to derive a simple formula relating environmental costs to per capita income, which could then be employed to calculate damages in countries where no relevant studies were available.

Estimates of damages based on meta-analysis have been provided in a formal sense in two studies carried out in the US on recreation demand (Smith and Kaoru, 1990, Walsh, Johnson and McKean, 1989), and on air pollution (Smith and Huang, 1991). The results in the recreation studies indicate that, as one would expect, the nature of the site is significant for the WTP attached to a visit, as are the costs of substitutes and the opportunity cost of time. Choice of functional form in the estimating equations also appears to play a part. In the air pollution study referred to above, it was found that damages per unit of concentration vary inversely with the average price of property in the study (the higher the price the lower the unit value of damage). If correct, it would enable an adjustment to the estimated value to be made on the basis of the average prices of properties in the area being investigated. However, the authors are cautious about the validity of the estimates obtained.

A formal meta-analysis is difficult to carry out, and for the most part has been beyond the scope of the ExternE Project. However, some of the 'expert' adjustments do make an informal meta analysis. For example, adjusting estimates of damages for size of population to obtain a per capita estimate and transferring that to the new study implicitly assumes that damages are proportional to population. Such adjustments are frequently made.

3.4.4 Conclusions on Benefit Transfer

Transferability depends on being able to use a large body of data from different studies and estimating the systematic factors that would result in variations in the estimates. In most cases the range of studies available are few. More can be done to carry out meta-analysis of the type indicated, but it will take time. The best practice in the meantime is to use estimates from sources as close to the one in which they are being applied and adjust them for differences in underlying variables where that is possible. Often the most important obstacle to systematic benefit transfer, however, is a lack of documentation in the existing valuation studies.

It is important to note that national boundaries themselves are not of any relevance in transferring estimates, except that there may be cultural differences that will influence factors such as frequency with which a person visits a doctor, or how he perceives a loss of visibility. In this sense there is no reason why the ExternE Project should not draw on the findings of the companion US Department of Energy study and other work, or transfer estimates from one country to another within Europe, as long as the above consideration is taken into account.

3.5 The Treatment of Discounting

3.5.1 Introduction

Discounting is the practice of placing lower numerical values on future benefits and costs as compared to present benefits and costs. In the context of the fuel cycle study it is an important issue because many of the environmental damages of present actions will occur many years from now and the higher the discount rate, the lower the value that will be attached to these damages. This can have major implications for policy.

This Section reviews the arguments for and against different discount rates, including the case for discounting environmental damages at a different rate from other projects. It offers some recommendations for the use of discount rates in the project context.

3.5.2 The Rationale for Discounting and Choice of Discount Rate

The practice of *discounting* arises because individuals attach less weight to a benefit or cost in the future than they do to a benefit or cost now. Impatience, or ‘time preference’, is one reason why the present is preferred to the future. The second reason is that, since capital is productive, an ECU’s worth of resources now will generate more than an ECU’s worth of goods and services in the future. Hence an entrepreneur would be willing to pay more than one ECU in the future to acquire an ECU’s worth of these resources now. This argument for discounting is referred to as the ‘marginal productivity of capital’ argument; the use of the word marginal indicates that it is the productivity of additional units of capital that is relevant.

If a form of damage, valued at ECU X today, but which will occur in T years time is to be discounted at a rate of r percent, the value of X is reduced to:

$$X/(1+r)^T.$$

Clearly the higher r and T are, the lower the value of the discounted damages. On the rationality of time discounting *per se* see Jevons (1871), Bohm-Bawerk (1884), Ramsey (1929), Pigou (1932). Typically discount rates in EC countries run at around 5-7% in real terms. The latter means that when future values are being computed, no allowance is made for general inflation of money values, and all damages are calculated in present prices. In the UK, the Treasury guidelines recommend a discount rate of 6% for public sector projects, but with a 3% rate for forestry projects, and with some relaxation of the 6% rule for other projects which have very long term effects. There are also special provisions for discounting decommissioning costs in connection with nuclear plants, where a discount rate of 2% is used just for that item. Although there is a general tendency to use a single rate of around 6% there are exceptions in government policy to allow for items where there are very long term impacts.

Excluding questions of risk, which are discussed later, the two main contenders for the choice of the social discount rate are the *social rate of time preference*, based on the rate of time preference, and the *opportunity cost of capital*, based on the marginal productivity of capital. While the two rates would be equal if there were efficient markets and no taxes, in practice time preference rates tend to be below the opportunity cost of capital. The early debate on which rate to use focused on the *sources* of the funds which were applied to the project and on the eventual *uses* of the benefits of the project (Marglin 1967; Feldstein ,1972; Lind, 1982). In particular, the extent to which the costs and benefits detracted from and added to consumption relative to savings was seen to be of key importance in determining the choice of the discount rate.

3.5.3 The Discounting Debate from an Environmental Perspective

The relationship between environmental concerns and the social discount rate operates in two directions. In analysing the first, one re-examines the rationale for discounting and the methods of calculating discount rates, paying particular attention to the problem of the environment. In the second, one looks at particular environmental concerns, and analyses their implications given different discount rates. Beginning with the first, the objections to the arguments for discounting can be presented under five headings:

a) pure time preference;
b) social rate of time preference;
c) opportunity cost of capital;
d) risk and uncertainty;
e) the interests of future generations.

Much of the environmental literature argues against discounting *in general* and high discount rates in particular (Parfit, 1983; Goodin, 1986). There is in fact no unique relationship between high discount rates and environmental deterioration. High rates may well shift the cost burden to future generations but, as the discount rate rises, so falls the overall level of investment, thus slowing the pace of economic development in general. Since natural resources are required for investment, the demand for such resources is lower at higher discount rates. High discount rates may also discourage development projects that compete with existing environmentally benign uses, e.g., watershed development as opposed to existing wilderness use. Exactly how the choice of discount rate impacts on the overall profile of natural resource and environment use is thus ambiguous. This point is important because it indicates the invalidity of the more simplistic generalisations that discount rates should be lowered to accommodate environmental considerations. This prescription has been challenged at an intuitive level by Krutilla (1967). For further discussions see Markandya and Pearce (1988) and Krautkraemer (1988).

3.5.4 Pure Individual Time Preference

In terms of *personal* preferences, no one appears to deny the impatience principle and its implication of a positive individual discount rate. However, arguments exist against permitting pure time preference to influence *social* discount rates, i.e., the rates used in connection with collective decisions. These can be summarised as follows. First, individual time preference is not consistent with individual lifetime welfare maximisation. This is a variant of a more general view than time discounting because impatience is irrational (see Strotz, 1956, and, Krutilla and Fisher, 1975). Second, what individuals want carries no necessary implications for public policy. Many countries, for instance, compulsorily force savings behaviour on individuals, e.g., through state pensions, indicating that the state overrides private preferences concerning savings behaviour. Third, the underlying value judgement is improperly expressed. A society that elevates want satisfaction to a high status should recognise that it is the satisfaction of wants *as they arise* that matters (see Goodin, 1986). But this means that it is tomorrow's satisfaction that matters, not today's assessment of tomorrow's satisfaction.

How valid these objections are to using pure time preference is debatable. Overturning the basic value judgement underlying the liberal economic tradition, i.e., that individual preferences should count for social decisions, requires good reason. Although strong arguments for paternalism do exist, they are not, in the author's view, sufficient to justify its use in this context. Philosophically the third argument, that the basic value judgement needs re-expressing, is impressive. In practical terms, however, the immediacy of wants in many developing countries where environmental problems are serious might favour the retention of the usual formulation of this basic judgement.

3.5.5 Social Rate of Time Preference

The social time preference rate attempts to measure the rate at which social welfare or utility of consumption falls over time. Clearly this will depend on the rate of pure time preference, on how fast consumption grows and, in turn, on how fast utility falls as consumption grows. It can be shown that the social rate of time preference is:

3.5.6 Opportunity Cost of Capital

The opportunity cost of capital is obtained by looking at the rate of return on the best investment of similar risk that is displaced as a result of the particular project being undertaken. It is only reasonable to require the investment undertaken to yield a return at least as high as that on the alternative use of funds. This is the basic justification for an opportunity cost discount rate. In developing countries where there is a shortage of capital, such rates tend to be very high and their use is often justified on the grounds of the allocation of scarce capital.[4]

The environmental literature has made some attempts to discredit discounting on opportunity cost grounds (Parfit, 1983; Goodin, 1986). The first criticism is that opportunity cost discounting implies a reinvestment of benefits at the opportunity cost rate, and this is often invalid. For example, at a 10% discount rate ECU 100 today is comparable to ECU 121 in two years time if the ECU 100 is invested for one year to yield ECU 10 of return and then both the original capital and the return are invested for another year to obtain a total of ECU 121. Now, if the return is consumed but not reinvested then, the critics argue, the consumption flows have no opportunity cost. What, they ask, is the relevance of a discount rate based on assumed reinvested profits if in fact the profits are consumed?

The second environmental critique of opportunity cost discounting relates to compensation across generations. Suppose an investment today would cause environmental damages of ECU X, T years from now. The argument for representing this damage in discounted terms by the amount ECU $X/(i+r)^T$ is the following. If this latter amount were invested at the opportunity cost of capital discount rate r, it would amount to ECU X in T years time. This could then be used to compensate those who suffer the damages in that year. Parfit argues, however, that using the discounted value is only legitimate if the compensation is *actually* paid. Otherwise, he argues, we cannot represent those damages by a discounted cost.[5]
The problem here is that actual and 'potential' compensation are being confused. The fact that there is a sum generated by the project that could be used for the *potential* compensation of the victim is enough to ensure its efficiency. Whether the compensation should *actually* be carried out is a separate question and one which is not relevant to the issue of how to choose a discount rate.

[4] It should be noted, however, that the allocation of scarce capital can be achieved without making adjustments to the discount rate. What is required is that a premium be attached to capital such that each dollar invested has a value greater than one dollar in the project calculations. Examples of how this might work are given in Markandya and Pearce (1988).

[5] See Parfit (1983). Along similar grounds, but more extreme, is the position that calculating present values in which the allocations which accrue to *different* people are discounted and added up is invalid. This is because the latter is a distributional issue and cannot be addressed through the use of a discount rate. However, taking this to its logical conclusion would also imply that we could not add up benefits or costs across individuals even at a point in time, which is a familiar distributional difficulty in cost benefit analysis but one that can and has been addressed in a number of ways.

These two arguments against opportunity cost discounting are not persuasive, although the first can be argued to be relevant to using a weighted average of the opportunity cost and the rate of time preference. In practice the rates of discount implied by the opportunity cost are within the range of discount rates actually applied to projects in the EC countries. In the UK for example, the real returns to equity capital are in the range of 5-7%, which is consistent with the Treasury guidelines of the discount rate that should be used for public sector project discounting.

3.5.7 Risk and Uncertainty

It is widely accepted that a benefit or cost should be valued less, the more uncertain is its occurrence. The types of uncertainty that are generally regarded as being relevant to discounting are:

- uncertainty about whether an individual will be alive at some future date (the 'risk of death' argument),
- uncertainty about the preferences of the individual in the future, and
- uncertainty about the size of the benefit or cost.

The risk of death argument is often used as a rationale for the impatience principle itself, the argument being that a preference for consumption now rather than in the future is partly based on the fact that one may not be alive in the future to enjoy the benefits of ones restraint. The argument against this is that although an individual may be mortal, 'society' is not and so its decisions should not be guided by the same consideration. This is another variant of the view that, in calculating social time preference rates, the pure time preference element (z) may be too high.

Second, uncertainty about preferences is relevant to certain goods and perhaps even certain aspects of environmental conservation. However, economists generally accept that the way to allow for uncertainty about preferences is to include *option value* in an estimate of the benefit or cost rather than to increase the discount rate (see Section B of this report for a discussion of Option Values in the context of damage estimation).

The third kind of uncertainty is relevant, but the difficulty is in allowing for it by adjusting the discount rate. Such adjustments assume that the scale of risks is increasing exponentially over time. Since there is no reason to believe that the risk factor takes this particular form, it is inappropriate to correct for such risks by raising the discount rate. This argument is in fact accepted by economists, but the practice of using risk-adjusted discount rates is still quite common among policy makers. For example there is a 2% 'premium' attached to the officially recommended 5% 'test discount rate' in the United Kingdom in the presence of 'benefit optimism' (UK Treasury, 1980)[6].

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[6] Although, the UK Treasury guidelines recommend a discount rate of 6% for public sector projects, an excpetion is made for forestry projects, where a 3% rate is applied. There are also special provisions for discounting decommissioning costs in connection with nuclear plants, where a discount rate of 2% is used just for that item. Thus although there is a general tendency to use a single rate of around 6% there are exceptions in government policy to allow for items where there are very long term impacts

If uncertainty is not to be handled by discount rate adjustments then how should it be treated? The alternative is to make adjustments to the underlying cost and benefit streams. This involves essentially replacing each uncertain benefit or cost by its *certainty equivalent*. This procedure is theoretically correct, but the calculations involved are complex and it is not clear how operational the method is. However, this does not imply that adding a risk premium to the discount rate is the solution because, as has been shown, the use of such a premium *implies* the existence of *arbitrary certainty equivalents* for each of the costs and benefits.

3.5.8 The Interests of Future Generations

The extent to which the interests of future generations are safeguarded when using positive discount rates is a matter of debate within the literature. With overlapping generations, borrowing and lending can arise as some individuals save for their retirement and others dissave to finance consumption. In such models, it has been shown that the discount rate that emerges is not necessarily efficient, i.e., it is not the one that takes the economy on a long run welfare maximising path. These models, however, have no ‘altruism’ in them. Altruism is said to exist when the utility of the current generation is influenced not only by its own consumption, but also by the utility of future generations. This is modelled by assuming that the current generation’s utility (i), is also influenced by the utility of the second generation (j) and the third generation (k). This approach goes some way towards addressing the question of future generations, but it does so in a rather specific way. Notice that what is being evaluated here is the current generation’s judgement about what the future generations will think is important. It does not therefore yield a discount rate reflecting some broader principle of the rights of future generations. The essential distinction is between generation (i) judging what generation (j) and (k) want (selfish altruism) and generation (i) engaging in resource use so as to leave (j) and (k) with the maximum scope for choosing what they want (disinterested altruism). On overlapping generations models see Diamond (1965). On altruism see Page (1977).

Although this form of altruism is recognised as important, its implications for the interest rate and the efficiency of that rate have yet to be worked out. The validity of this overlapping generations argument has also been questioned on the grounds of the ‘role’ played by individuals when they look at future generations’ interests. Individuals make decisions in two contexts, ‘private’ decisions reflecting their own interests and ‘public’ decisions in which they act with responsibility for fellow beings and for future generations. Market discount rates, it is argued, reflect the private context, whereas social discount rates should reflect the public context. This is what Sen calls the ‘dual role’ rationale for social discount rates being below the market rates. It is also similar to the ‘assurance’ argument, namely that people will behave differently if they can be assured that their own action will be accompanied by similar actions by others. Thus, we might each be willing to make transfers to future generations only if we are individually assured that others will do the same. The ‘assured’ discount rate arising from collective action is lower than the ‘unassured’ rate (Becker, 1988; Sen, 1982).

There are other arguments that are used to justify the idea that market rates will be 'too high' in the context of future generations' interests. The first is what Sen calls the 'super responsibility' argument (see Sen, 1982). Market discount rates arise from the behaviour of individuals, but the state is a separate entity with the responsibility for guarding collective welfare and the welfare of future generations. Thus the rate of discount relevant to state investments will not be the same as the private rate and, since high rates discriminate against future generations, we would expect the state discount rate to be lower than the market rate.

The final argument used to justify the inequality of the market and social rates is the 'isolation paradox'. The effect of this is rather similar to that generated by the assurance problem but it arises from slightly different considerations. In particular, when individuals cannot capture the entire benefits of present investments for their own descendants, the private rate of discount will be below the social rate (Sen, 1961, and Sen 1967).

Hence, for a variety of reasons relating to future generations' interests, the social discount rate may be below the market rate. The implications for the choice of the discount rate are that there is a need to look at an individual's 'public role' behaviour, or to leave the choice of the discount rate to the state, or to try and select a rate based on a collective savings contract. However, none of these options appears to offer a practical procedure for determining the discount rate in quantitative terms. What they do suggest is that market rates will not be proper guides to social discount rates once future generations' interests are incorporated into the social decision rule. These arguments can be used to reject the use of a market based rate *if it is thought that the burden of accounting for future generations' interests should fall on the discount rate*. However, this is a complex and almost certainly untenable procedure. It may be better to define the rights of future generations and use these to circumscribe the overall evaluation, leaving the choice of the discount rate to the conventional current-generation-oriented considerations. Such an approach is illustrated shortly.

3.5.9 Discount Rates and Irreversible Damage

So far the rationale for discounting and the debate on the choice of the discount rate has been looked at from an environmental perspective. However, one specific issue that might, *prima facie,* imply the adjustment of the discount rate is that of irreversible damage. This is examined below.

As the term implies the concern is with decisions that cannot be reversed, such as the flooding of a valley, the destruction of ancient monuments, radioactive waste disposal, tropical forest loss and so on. One approach which incorporates these considerations into a cost-benefit methodology is that developed by Krutilla and Fisher (1975) and generalised by Porter (1982).

Consider a valley containing a unique wilderness area where a hydroelectric development is being proposed. The area, once flooded, would be lost forever. The resultant foregone benefits are clearly part of the costs of the project. The net development benefits can then be written as:

$$\text{Net Benefit} = B(D) - C(D) - B(P)$$

where B(D) are the benefits of development (the power generated and/or the irrigation gained), C(D) are the development costs and B(P) are the net benefits of preservation (i.e., net of any preservation costs). All the benefits and costs need to be expressed in present value terms. The irreversible loss of the preservation benefits might suggest that the discount rate should be set very low since it would have the effect of making B(P) relatively large because the preservation benefits extend over an indefinite future. Since the development benefits are only over a finite period (say 50 years) the impact of lowering the discount rate is to lower the net benefits of the project. However, in the Krutilla-Fisher approach the discount rate is not adjusted. It is treated 'conventionally', i.e. set equal to some measure of the opportunity cost of capital.

Instead of adjusting the discount rate in this way Krutilla and Fisher note that the value of benefits from a wilderness area will grow over time. The reasons for this are that: (a) the supply of such areas is shrinking, (b) the demand for their amenities is growing with income and population growth and (c) the demand to have such areas preserved even by those who do not intend to use them is growing (i.e., what are referred to as 'existence values' are increasing). The net effect is to raise the 'price' of the wilderness at some rate of growth per annum, say g%. However, if the price is growing at a rate of g% and a discount rate r% is applied to it, this is equivalent to holding the price constant and discounting the benefit at a rate (r-g)%. The adjustment is very similar to lowering the discount rate but it has the attraction that the procedure cannot be criticised for distorting resource allocation in the economy by using variable discount rates.

Krutilla and Fisher engage in a similar but reverse adjustment for development benefits. They argue that technological change will tend to reduce the benefits from developments such as hydropower because superior electricity generating technologies will take their place over time. The basis for this argument is less clear but, if one accepts it, then the development benefits are subject to technological depreciation. Assume this rate of depreciation is k%. Then the effect is to produce a net discount rate of (r+k)%, thereby lowering the discounted value of the development benefits.

3.5.10 A Sustainability Approach

The environmental debate has undoubtedly contributed to the rationale for discounting. But it has not been successful in demonstrating a case for rejecting discounting as such. This discussion began by examining the concern over the use of discount rates which reflect pure time preference, but concluded that this concern does not provide a case for rejecting pure time preference completely. However, it was noted that an abnormally high time preference rate can be generated when incomes are falling and when environmental degradation is taking place. In these circumstances, it is inappropriate to evaluate policies, particularly environmentally relevant ones, with discount rates based on these high rates of time preference.

The arguments by environmentalists against the use of opportunity cost of capital discount rates were also, in general, not found to be persuasive. It was also observed that, to account for uncertainty in investment appraisal, it was better to adjust the cost and benefit streams for the uncertainty rather than to add a 'risk premium' onto the discount rate.

Finally, under the general re-analysis of the rationale for discounting, the arguments for adjusting discount rates on various grounds of inter-generational justice were examined. Although many of these arguments have merit, it was concluded that adjusting the discount rate to allow for them was not, in general, a practicable or efficient procedure. However, the need to protect the interests of future generations remains paramount in the environmental critique of discounting. Some alternative policy is therefore required if the discount rate adjustment route is not to be followed. One approach is through a ‘sustainability constraint’.

The notion of ‘sustainability’, or ‘sustainable development’ is widely discussed in the 1987 report of the World Commission on Environment and Development (the ‘Bruntland Commission’). While few attempts have been made to analyse the concept rigorously the basic idea is that economic development requires a strong protective policy towards the natural resource base. One might say that the resource base should be maintained intact in some sense, or even enhanced. The link between maintaining the *overall* capital base of the economy, man-made *and* ‘natural capital’, and inter-generational equity is established in some of the recent literature (Pearce, 1993). Sustainability advocates go further and separate out natural capital for special attention. In the developing world one justification for this would be the close dependence of major parts of the population on natural capital (soil, water and biomass). More generally, ecological science suggests that much natural capital cannot be substituted for by man-made capital (an example might be the ozone layer).[7]

If conservation of natural environments is a condition of sustainability, and if sustainability meets many (perhaps all) of the valid criticisms of discounting, how might it be built into project appraisal? Requiring that no project should contribute to environmental deterioration would be absurd. But requiring that the overall *portfolio* of projects should not contribute to environmental deterioration is not absurd. One way to meet the sustainability condition is to require that any environmental damage be *compensated* by projects specifically designed to improve the environment. The sustainability approach has some interesting implications for project appraisal. One of these is that the choice of discount rates becomes less relevant. The goal of adjusting discount rates to capture environmental effects is better served by the sustainability condition. If this is correct, the sustainability condition deserves more investigation. Although it may have quite radical implications, it offers the prospect of avoiding belabouring the ‘tyranny of discounting’ and of asking that all ethical and environmental concerns be accounted for by discount rate adjustment (as an example of this approach, see Pearce, Barbier and Markandya, 1989). However, it is important to note that such an investigation lies outside the scope of the ExternE Study.

To some extent, a ‘sustainability’ approach is already followed, when environmentalists have argued that, in some key cases, protection for key resources and environments has to be guaranteed, *irrespective* of whether it can be justified on cost-benefit grounds at conventional discount rates.

[7] The concept of sustainability has been analysed in Pezzey (1989), and Pearce, Barbier and Markandya (1989). For a discussion of natural and man-made capital in optimal intertemporal models see Solow (1986).

Although there are merits in favour of such an argument, what is being called for here is more than that. What is needed is a *systematic procedure* by which a sustainability criterion can be invoked in support of certain actions. Such a procedure does not exist, but it would be desirable to develop one.

3.5.11 The Use of a Constant Discount Rate

Although there is not presumption that the discount rate should be constant, most analysts take a constant discount rate. In the earlier ExternE work this was indeed the case. The 'central rate taken was 3 percent, with a sensitivity analysis carried out for rates of 0 and 10 percent. More recently, however, the literature has developed and argued that the rate should not be constant. In a recent paper looking at how individuals value present versus future risks, Cropper et al. (1994) estimate a nominal annual rate of around 16.8%. Most importantly, however, the Cropper paper points to a **declining** rate of discount – i.e. to a rate that is not constant over time but gets smaller as the time horizon lengthens. So whereas the 16.8% rate applies to a 5 year time horizon, a rate of only 3.4% applies to a 100 year horizon. The key point that this and other papers are coming up with is that the discount rate for very long-term impacts needs to decline. This is relevant for fuel cycles such as the nuclear, where there are indeed some impacts that are far into the future. Although the issue of time variant discount rates has been discussed in the present work, it has not been adopted as part of the ExternE methodology.

3.5.12 Conclusions and Recommendations

This Section has reviewed the arguments for different discount rates and concluded that:

- the arguments against discounting are not valid;
- a social time preference rate of around 2-4% would be justified on the grounds of incorporating a sustainable rate of per capita growth and an acceptable rate of time preference;
- rates of discount based on the opportunity cost of capital would lie at around 5-7% for the EC countries. There are arguments to suggest that these may be too high on social grounds. It is important to note that these arguments are not specific to environmental problems;
- the treatment of uncertainty is better dealt with using other methods, than modifying the discount rate;
- where irreversible damages are incurred, it is better to allow for these by adjusting the values of future costs and benefits than by employing a lower discount rate specifically for that project or component;
- for projects with latent impacts over a long period a declining discount rate appears to be appropriate.
- for projects where future damage is difficult to value, and where there could be a loss of natural resources with critical environmental functions, a ‘sustainability’ approach is recommended. This implies debiting the activity that is causing the damage with the full cost of repairing it, irrespective of whether the latter is justified.

As noted above, for the ExternE study it was recommended that the lower of the above time preference rates be employed for discounting future damages. A figure of 3% would be a central acceptable rate. In addition, appropriate increases in future values of damages to allow for increased demands for environmental services in the face of a limited supply of such facilities should be made. Finally, when (and only when) the fuel cycle can be identified with damages to the environment that would affect the *sustainability* of the ecosystem, a full repairing cost should be debited to that component. It should be noted that, in some cases, the damage estimates have also been computed for rates of 0% and 10%. The range obtained provides an indication of the sensitivity of damage estimation to discounting. It is acknowledged that a 10% rate is excessive, but has been applied simply to demonstrate the effect of discounting at commercial rates.

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4. MODELS FOR AIR POLLUTION ANALYSIS

4.1 Introduction

Given increased understanding of the importance of long range transboundary transport of airborne pollutants, there was an obvious need in the project for a harmonised European-wide database supporting the assessment of environmental impacts from air pollution. In the very beginning of the ExternE Project, work focused on the assessment of local scale impacts, and teams from different countries made use of the data sources available in each country. Country specific data sources and grid systems were not compatible when extending analysis to the European scale, so it was logical to set up a common European-wide database by using official sources like EUROSTAT and then making this available to all ExternE teams. The next step was to establish a link between the database and all the models required for the assessment of external costs to guarantee a harmonised and standardised implementation of the theoretical methodological framework. This led to the development of the EcoSense model.

The objectives for the development of EcoSense were:

- to provide a tool supporting a standardised calculation of fuel cycle externalities,
- to integrate relevant models into a single system,
- to provide a comprehensive set of relevant input data for the whole of Europe,
- to enable the transparent presentation of intermediate and final results, and
- to support easy modification of assumptions for sensitivity analysis.

As health and environmental impact assessment is a field of large uncertainties and incomplete, but rapidly growing understanding of the physical, chemical and biological mechanisms of action, it was a crucial requirement for the development of the EcoSense system to allow an easy integration of new scientific findings. As a consequence, all the calculation modules (except for the ISC-model, see below) are designed in a way that they are a *model-interpreter* rather than a *model*. Model specifications like e.g. chemical equations, dose-response functions or monetary values are stored in the database and can be modified by the user. This concept allows easy modification of model parameters, and avoids the problems of 'black box' systems by allowing the user to track stepwise through the analysis.

4.2 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.

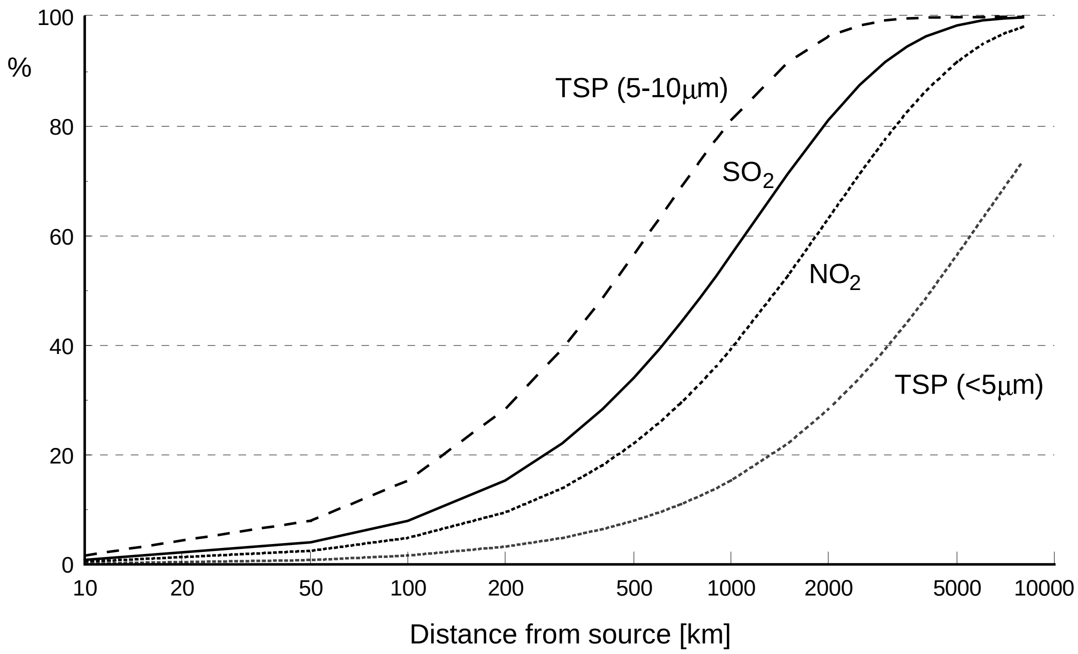
The atmospheric pollutant transport processes we have modelled in our analysis of fossil fuel cycles can be classified into three groups. These are separated according to their chemical characteristics and the atmospheric chemical and physical processes involved in their formation. They are:

- Primary pollutants directly emitted from the stack. These include particulate matter and sulphur dioxide (SO_2);
- Secondary sulphur and nitrogen species formed from the primary emissions of SO_2 and NO_x. 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, is 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 of 100 km of the source.

In one of our earlier reports (European Commission, 1995) it was estimated that assessment over a range of 1000 km or more was necessary to capture 80% or more of the damages linked to emission of NO_x, SO_2, and fine particles (Figure 4.1). 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 a receptor-orientated Lagrangian trajectory model. 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, and is described separately in this report (Chapter 16).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TSP (<5μm)

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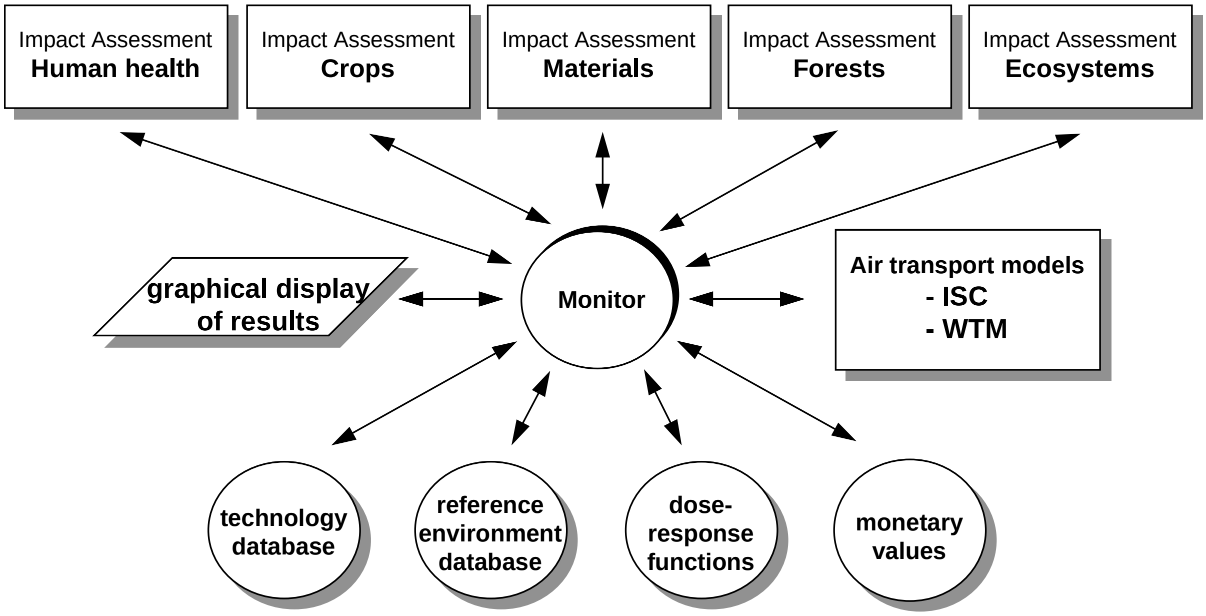

Figure 4.2 Structure of the EcoSense model

4.4.1 Reference Technology Database

The reference technology database holds a small set of technical data describing the emission source (power plant) that are mainly related to air quality modelling, including e.g. emission factors, flue gas characteristics, stack geometry and the geographic co-ordinates of the site.

4.4.2 Reference Environment Database

The reference environment database is the core element of the EcoSense database, providing data on the distribution of receptors, meteorology as well as a European wide emission inventory. All geographical information is organised using the EUROGRID co-ordinate system, which defines equal-area projection gridcells of 10 000 km^2 and 100 km^2 (Bonnefous and Despres, 1989), covering all EU and European non-EU countries.

Data on population distribution and crop production are taken from the EUROSTAT REGIO database, updated in part using information from national statistics. The material inventories are quantified in terms of the exposed material area from estimates of 'building identikits' (representative buildings). Surveys of materials used in the buildings in some European cities were used to take into account the use of different types of building materials around Europe. Critical load maps for nitrogen deposition are available for nine classes of different ecosystems, ranging from Mediterranean scrub over alpine meadows to tundra areas. To simplify access to the receptor data, an interface presents all data according to administrative units (e.g. country, state) following the EUROSTAT NUTS classification scheme. The system automatically transfers data between the grid system and the respective administrative units.

In addition to the receptor data, the reference environment database provides elevation data for the whole of Europe on the 10x10 km grid, which is required to run the Gaussian plume model, as well as meteorological data (precipitation, wind speed and wind direction) and a European-wide emission inventory for SO_2, NO_x and NH_3 from EMEP 1990 which has been transferred to the EUROGRID-format.

4.4.3 Exposure-Response Functions

Using an interactive interface, the user can define any exposure-effect model as a mathematical expression. The user-defined function is stored as a string in the database, which is interpreted by the respective impact assessment module at runtime. All exposure-response functions compiled by the various ‘area experts’ of the ExternE Maintenance Project are stored in the database.

4.4.4 Monetary Values

The database provides monetary values for most of the impact categories following the recommendations of the ExternE economic valuation task group. In some cases there are alternative values to carry out sensitivity analysis

4.4.5 Air Quality Models

To cover different pollutants and different scales, EcoSense provides two air transport models completely integrated into the system:

- The Industrial Source Complex Model (ISC) is a Gaussian plume model developed by the US-EPA (Brode and Wang, 1992). The ISC is used for transport modelling of primary air pollutants (SO_2, NO_x, particulates) on a local scale.
- The Windrose Trajectory Model (WTM) is a user-configurable trajectory model based on the windrose approach of the Harwell Trajectory Model developed at Harwell Laboratory, UK (Derwent, Dollard, Metcalfe, 1988). For current applications, the WTM is configured to resemble the atmospheric chemistry of the Harwell Trajectory Model. The WTM is used to estimate the concentration and deposition of acid species on a European wide scale.

All input data required to run the Windrose Trajectory Model are provided by the EcoSense database. A set of site specific meteorological data has to be added by the user to perform local scale modelling using the ISC model. The concentration and deposition fields calculated by the air quality models are stored in the reference environment database. Section 4.4 gives a more detailed description of the two models incorporated into EcoSense.

4.4.6 Impact Assessment Modules

The impact assessment modules calculate the physical impacts and - as far as possible - the resulting damage costs by applying the exposure-response functions selected by the user to each individual gridcell, taking into account the information on receptor distribution and concentration levels of air pollutants from the reference environment database. The assessment modules support the detailed step-by-step analysis for a single endpoint as well as a more automated analysis including a range of prespecified impact categories.

4.4.7 Presentation of Results

Input data as well as intermediate results can be presented on several steps of the impact pathway analysis in either numerical or graphical format. Geographical information like population distribution or concentration of pollutants can be presented as maps. EcoSense generates a formatted report with a detailed documentation of the final results that can be imported into a spreadsheet programme.

4.5 The Air Quality Models Integrated in EcoSense

4.5.1 Local Scale Modelling of Primary Pollutants - the Industrial Source Complex Model

Close to the plant, i.e. at distances of 10-50 km, chemical reactions in the atmosphere have little influence on the concentrations of primary pollutants, if NO and its oxidised counterpart NO_2 can be summarised as NO_x. Due to the large emission height on top of a tall stack, the near surface ambient concentrations of the pollutants at short distances from the stack are heavily dependent on the vertical mixing 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, the most economic way of assessing ambient air concentrations of primary pollutants on a local scale is a model which neglects chemical reactions but is detailed enough in the description of turbulent diffusion and vertical mixing. An often used model, which meets these requirements, is the Gaussian plume model. The concentration distribution from a continuous release into the atmosphere is assumed to have a Gaussian shape:

$$c(x,y,z) = \frac{Q}{u2\pi\sigma_y\sigma_z} \cdot \exp -\frac{y^2}{2\sigma_y{}^2} \;\cdot\; \exp -\frac{(z-h)^2}{2\sigma_z{}^2} + \exp -\frac{(z+h)^2}{2\sigma_z{}^2}$$

where:

where:	*c(x,y,z)*	concentration of pollutant at receptor location *(x,y,z)*
	Q	pollutant emission rate (mass per unit time)
	u	mean wind speed at release height
	σ_y	standard deviation of lateral concentration distribution at downwind distance *x*
	σ_z	standard deviation of vertical concentration distribution at downwind distance *x*
	h	plume height above terrain

The assumptions embodied into this type of model include those of idealised terrain and the3

$$h_{stab} = h + z_s - \min(z|_{(x,y)}, z_s + h_s)$$

where:

h	plume height, assuming flat terrain
h_s	height of the stack
z_s	height above mean sea level of the base of the stack
$z\|_{(x,y)}$	height above mean sea level of terrain at the receptor location

Mean terrain heights for each grid cell are provided by the reference environment database.

It is the responsibility of the user to provide the meteorological input data. These include wind direction, wind speed, stability class as well as mixing height, wind profile exponent, ambient air temperature and vertical temperature gradient.

4.5.2 Regional Scale Modelling of Primary Pollutants and Acid Deposition - the Windrose Trajectory Model

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 (see Chapter 16), the main species of interest in the regional assessments are the acidifying pollutants, formed from the primary emissions of SO_2 and NO_x. Both pollutants cause acid deposition, which has been studied in Western Europe over many years.

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.

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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).

The Windrose Trajectory Model (WTM) used in EcoSense to estimate the concentration and deposition of acid species on a regional scale was originally developed at Harwell Laboratory by Derwent and Nodop (1986) for atmospheric nitrogen species, and extended to include sulphur species by Derwent, Dollard and Metcalfe (1988). The model is a receptor-orientated

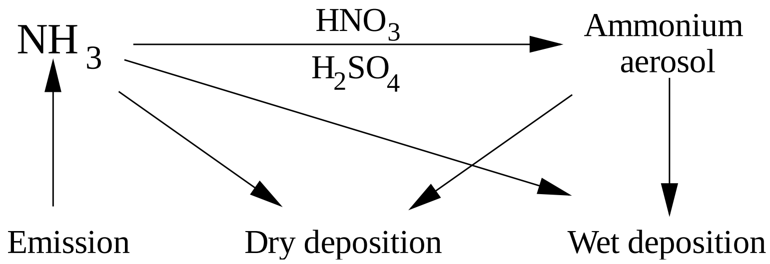

Figure 4.3 Chemical reactions of the sulphur and nitrogen species included in the Harwell Trajectory Model.

In EcoSense, the model is implemented by means of

- a set of parameters and chemical equations in the EcoSense database which defines the model
- a model interpreter (wmi.exe)
- a set of meteorological input data (gridded wind roses and precipitation fields) in the reference environment database
- emission inventories for NO_x, SO_2 and ammonia, which are also provided in the reference environment database
- additional emissions of the plant from the reference technology database

The 1990 meteorological data were provided by the Meteorological Synthesizing 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, receptor specific, the mean annual wind rose (frequency distribution of the wind per sector), the mean annual wind speed, and total annual precipitation. Base line emissions of NO_x, SO_2 and NH_3 for Europe are taken from the 1990 EMEP inventory (Sandnes and Styve, 1992).

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5. ASSESSMENT OF UNCERTAINTY

5.1 Introduction

Earlier analysis conducted in the ExternE Project determined that the uncertainties of external cost assessment were likely to be significant. However, it also determined that these uncertainties were difficult to quantify. This issue was developed as a specific task in the project over the period 1996 to 1997. Attention focused on air pollution effects as these proved to be so important in the 1995 series of reports.

The principal steps in the analysis of air pollution damages are as follows:

- specification of the relevant technologies and the environmental burdens they impose (e.g. kg/s of particulates emitted by a power plant);
- calculation of increased pollutant concentration in all affected regions (e.g. µg/m^3 of particulates, using models of atmospheric dispersion and chemistry);
- calculation of the resulting dose and the physical impacts (e.g. number of cases of asthma due to these particulates, using a dose-response function);
- economic valuation of these impacts (e.g. multiplication by the cost of a case of asthma).

Formally this can be represented as an equation for the incremental damage D and cost C due to an incremental quantity Q of a pollutant emitted by a source

$$C = p\,D = p \sum F_{dr,I}\,(c_i\,(Q)) \qquad (5.1)$$

where

$c_i(Q)$ = increase in pollutant concentration for receptor i, and
$F_{dr,i}(c)$ = dose-response function for receptor i;
p = unit cost ("price") for this damage.

The numbers are summed over all receptors (population, crops, buildings, ...) that may be affected by this pollutant and that one wants to include in the analysis.

One needs to identify and quantify the sources of uncertainty, and then combine them over the steps of the impact pathway, according to the rules of statistics. If one wants to treat arbitrary probability distributions, one needs a numerical Monte Carlo analysis. This is rigorous and appears to be the most generally recommended approach (Morgan and Henrion 1990). More recently Rowe et al (1995) have introduced a simplification by using beta distributions with four adjustable parameters, determined by a fit to expert opinions of low, median and high values and the corresponding probabilities. [8]

Quantifying the sources of uncertainty in this field is problematic because of the general lack of information. Usually one has to fall back on subjective judgement, preferably by the experts of the respective disciplines. An interesting project to elicit expert judgement of uncertainties has been carried out by Morgan et al (1984); it has also been summarised by Morgan and Henrion (1990). In this project experts were asked to try and specify the entire cumulative probability distribution, and these distributions were then fed into a Monte Carlo analysis.

In the present section we argue that it may not be necessary to worry about details of the probability distributions, because they wash out in the final result thanks to the central limit theorem, as pointed out recently by Slob (1994). Thus it suffices to specify geometric mean and geometric standard deviations, or equivalently, multiplicative confidence intervals about the geometric mean (which is usually quite close to the median). This can greatly simplify the expert judgement elicitation: multiplicative confidence intervals are much easier to specify than an entire probability distribution. Likewise it facilitates the interpretation of the final result.

This Chapter builds on the ideas of Slob (1994) and extends his argument to applications in public policy, where one needs to know the total damage summed over all receptors, not just the damage for a particular site or receptor. Such summation appears to violate the assumption of a multiplicative process. However, by examining the removal of a pollutant from the environment, there can also be significant cancellation of errors, especially if the receptor distribution is uniform. In fact, if the key parameters (receptor density, dose response function, and pollutant removal rate) are uniform, the expression for the total damage becomes multiplicative, after all. Curtiss and Rabl (1996) have shown, by means of extensive numerical calculations based on real data, that this simple multiplicative "uniform world model" is remarkably accurate for air pollution: it yields the correct damage estimate within a factor of three for a rather extreme range of conditions (from a source at a rural site on the Atlantic coast of France to one in Greater Paris, a metropolitan region with 20% of the population of France).

We discuss typical error distributions for the factors in the expression for the total damage and argue that a lognormal model for the result appears very plausible. We present explicit data for error distributions of two key parameters: the deposition velocity of atmospheric dispersion models, and the value of statistical life; they are close to lognormal.

[8] In passing we also mention a rather different approach, the NUSAP system proposed by Funtowicz and Ravetz (1990). This system assigns an elaborate set of qualitative attributes to several stages of the process of obtaining information. While the principle of such an assessment of information quality may be appealing, the complexity is forbidding in practice. Also, no methodology has been offered for aggregating NUSAP data over the steps of the impact pathway analysis.

To the extent that the distribution of the result is lognormal, the geometric mean equals the median and the geometric standard deviation has a simple interpretation in terms of multiplicative confidence intervals around the median.

5.2 Dose-response functions

5.2.1 The Form of the Dose-Response Function

By definition a dose-response function starts at the origin, and in most cases it increases monotonically with dose X, as sketched schematically in Figure 5.1. At very high doses the function may level off in S-shaped fashion, implying saturation. Dose-response functions for health are determined from epidemiological studies or from laboratory studies. Since the latter are mostly limited to animals, the extrapolation to humans introduces large uncertainties. Another major difficulty is that one needs relatively high doses in order to obtain observable non zero responses in a sample of realistic size; such doses are usually far in excess of the levels one is concerned with in environmental impact studies. Thus there is a serious problem of how to extrapolate from the observed data towards low doses. Figure 5.1 indicates several possibilities. The simplest is the linear model, i.e. a straight line from the origin through the observed data point(s). Many cancers, in particular from radioactivity, seem to follow this model.

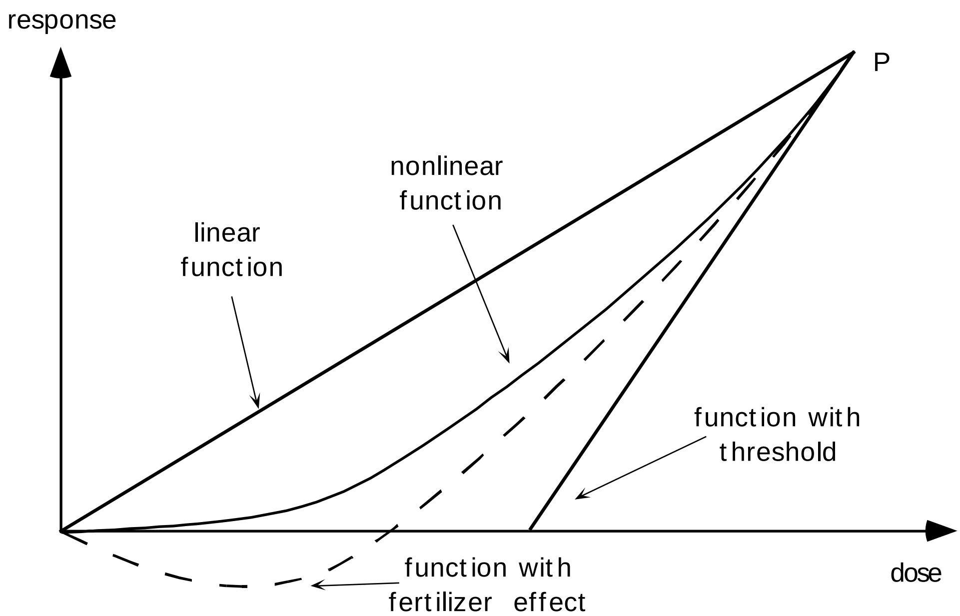

Figure 5.1 Possible behaviour of dose-response functions at low doses: the four functions shown have the same value at P.

Another possibility is the "hockey stick": a straight line down to some threshold, and zero effect below that threshold. Thresholds occur when an organism has a natural repair mechanism that can prevent or counteract damage up to a certain limit. Many dose-response functions for non-cancer toxicity are of this type.

There is even the possibility of a "fertiliser effect" at low doses, as indicated by the dashed line in Figure 5.1. This can be observed, for example, in the dose-response functions for the impact of NO_x and SO_X on crops: a low dose of these pollutants can increase the crop yield, in other words the damage is negative. Such a fertiliser effect can occur with pollutants that provide trace elements needed by an organism.

For air pollution the dose-response function is usually stated directly in terms of ambient concentration, hence the term exposure-response (E-R) function is more appropriate. In practice most dose-response functions used in ExternE, in particular all the ones for health, are linear, with slope

$$f_{E\text{-}R} = \frac{dF_{E-R}}{dc} \qquad (5.2)$$

In that case the equation for damage simplifies greatly.

For the calculation of incremental damage costs there is no difference between the linear and the hockey stick function (with the same slope), if the background concentration is everywhere above this threshold. For some air pollutants, e.g. particulates, the background in most industrialised countries is above the level where effects are known to occur (Dockery et al. 1993, Dockery and Pope 1994, Lipfert 1994). Thus the question of the precise form of the dose-response function at extremely low doses is irrelevant for these pollutants: whatever the threshold, if there is one, it is below the background concentrations of interest.

5.3 Summation over Receptors

5.3.1 Integration Over Areas Affected by Air Pollution

We focus on pollutants that are first emitted into the air, even if they may then be deposited onto soil, plants or water before damaging their receptors. In the latter case, in order to unify the presentation, we interpret the dose-response function as being aggregated over the steps from ambient concentration to final receptor (i.e. an effective dose-response function); this can usually be accomplished by means of suitably chosen multiplicative factors.

In the sum over receptors there can be significant cancellations, because the law of conservation of matter helps reduce the errors from the dispersion model.

This can be understood by considering a simplified example of an impact due to the deposition of a pollutant if the dose-response function is linear and the density of receptors uniform. In this limiting case the error due to the dispersion model is zero because overprediction at one site is exactly compensated by underprediction elsewhere, assuming that the analysis covers the entire geographic range over which the pollutant is deposited. Thus the net error due to dispersion models can be much smaller than commonly believed.

It will be convenient to write the damage D as an integral over land area by introducing $\rho(\mathbf{x})$, the density of receptors at point $\mathbf{x} = (x,y)$,

$$D = \int dx \int dy \quad \rho(\mathbf{x})\, F_{E\text{-}R}(\mathbf{x}, c(\mathbf{x},Q)) \tag{5.3}$$

where
$c(\mathbf{x},Q)$ = concentration increase at $\mathbf{x}$ due to emission Q,
and the notation $F_{E\text{-}R}(\mathbf{x}, c(\mathbf{x},Q))$ allows for the possibility that the exposure-response function may depend on local conditions such as the background concentration of a pollutant. Equation 5.3 describes the damage due to a steady state point source; it can readily be extended to a set of sources, e.g. line or area sources, or to time varying emissions.

5.3.2 Marginal Damage

From here on we limit ourselves to the important case of a linear exposure-response function $F_{E\text{-}R}(\mathbf{x}, c(\mathbf{x},Q))$

$$F_{E\text{-}R}(\mathbf{x}, c(\mathbf{x},Q)) = f_{E\text{-}R}\, c(\mathbf{x}) \tag{5.4}$$

where $f_{E\text{-}R}$ is the slope of the dose-response function. With that assumption one can write Equation 5.3 for the damage in the form

$$D = f_{E\text{-}R} \int dx \int dy \quad \rho(\mathbf{x})\, c(\mathbf{x}) \tag{5.5}$$

This is obviously exact for any pollutant whose dose-response function is linear, or a straight line with a threshold that is everywhere below the background. It is also valid, regardless of dose-response function, for the evaluation of any marginal impacts, i.e. impacts from small pollutant increments because in that case one can linearise the dose-response function. Since for primary pollutants $c(\mathbf{x})$ is linear in the emission, it follows that Equation 5.5, and the remainder of this report, is equally applicable to steady state situations and to emissions or damages that vary with time.

It will be interesting to relate the concentration c(**x**) to the removal rate of the pollutant. There are essentially three mechanisms by which an air pollutant can disappear from the atmosphere (Seinfeld 1986):

1) dry deposition (uptake at the earth's surface by soil, water or vegetation)
2) wet deposition (absorption into droplets followed by droplet removal by precipitation),
3) decay or transformations (e.g. decay of radionuclides, or chemical transformation of SO_2 to $(NH_4)_2SO_4$).

When evaluating the damage of the original pollutant, this pollutant is no longer counted in the equation once it has been transformed; rather from that point on a different dose-response function comes into play for the secondary pollutant.

The dry deposition rate is proportional to the concentration c(**x**) at the earth's surface, and it is customarily written in the form

$$F_{dry}(\mathbf{x}) = v_{dry}\, c(\mathbf{x}) \tag{5.6}$$

where

$F_{dry}(\mathbf{x})$ = deposition flux (in $kg/m^2{\cdot}s$), and
v_{dry} = dry deposition velocity (m/s).

Wet deposition and decay or transformation can likewise be characterised in terms of fluxes $F_{wet}(\mathbf{x})$ and $F_{trans}(\mathbf{x})$, defined as the rate at which the pollutant is removed by these mechanisms per m^2 and per s. Even though in general these fluxes are not proportional to the surface concentration but rather to the average concentration in the air column above **x**, we can write the total removal flux

$$F(\mathbf{x}) = F_{dry}(\mathbf{x}) + F_{wet}(\mathbf{x}) + F_{trans}(\mathbf{x}) \tag{5.7}$$

in terms of the surface concentration c(**x**) as

$$F(\mathbf{x}) = k(\mathbf{x})\; c(\mathbf{x}) \tag{5.8}$$

if we allow the proportionality constant k(**x**) to vary with **x**. the units of k are m/s, like a velocity. Using F(**x**) and k(**x**) we can write the damage in the form

$$D = f_{E\text{-}R} \int dx \int dy \quad \rho(\mathbf{x})\, F(\mathbf{x})/k(\mathbf{x}) \tag{5.9}$$

This equation is exact if we interpret Equation 5.8 as the definition of k(**x**). The units of k(**x**) are m/s and we refer to it as "removal velocity", by analogy to the dry deposition velocity.

5.3.3 Uniform World Model

If the world were homogeneous, in the sense of uniform receptor density $\rho(\mathbf{x}) = \rho_{uni}$, and uniform removal velocity $k(\mathbf{x}) = k_{uni}$, the integral in Equation 5.9 would be simply

$$D = D_{uni} = f_{E\text{-}R}\, \rho_{uni}\, Q/k_{uni} \qquad (5.10)$$

because the surface integral of the removal flux equals the emission

$$Q = \int dx \int dy \quad F(\mathbf{x}) \qquad (5.11)$$

by conservation of matter.

Even though the assumption $k(\mathbf{x}) = k_{uni}$ may not appear very realistic, especially near a point source, the sensitivity to deviations from uniformity turns out to be surprisingly small, as Curtiss and Rabl (1996) have demonstrated, with detailed numerical calculations based on real receptor data and data of the EMEP model for long range atmospheric dispersion (Eliassen and Saltbones 1983, Barrett 1992, Sandnes 1993). This is shown in Figure 5.2. The reason is that for typical values of atmospheric dispersion parameters the total impact is dominated by regions sufficiently far from the source that the pollutant can be considered to be vertically well mixed in the planetary boundary layer, at least as far as expectation values are concerned.

The same approach can be used for the damage due to a secondary pollutant (Curtiss and Rabl 1996). The multiplicative essence of the equations remains. Now there is an additional factor: the rate $k_{1\text{-}2}$ at which the primary pollutant is transformed into the secondary; this rate is defined analogous to Equation 5.8 in units of m/s. In particular for the uniform world model we obtain as generalisation of Equation 5.10

$$D_{2,uni} = f_{E\text{-}R2}\, \rho_{2,uni}\, Q\, k_{1\text{-}2,uni}/(k_{1,uni}\, k_{2,uni}) \qquad (5.12)$$

where the subscript 2 indicates that concentration, dose-response function and damage refer to the secondary pollutant.

Thus the simple Equations 5.10 and 5.12 can be a useful first estimate and an indication of functional relationships. Details of atmospheric dispersion do not matter very much. It is intuitively plausible that the damage is proportional to the slope $f_{E\text{-}R}$ of the exposure-response function, to the density ρ of receptors and to the emission rate Q. Furthermore, it is inversely proportional to the removal velocity k. If there were no removal mechanism, the pollutant concentration would increase without limit and the damage would be infinite.

5.4 Combination of Uncertainties

5.4.1 Uncertainty of a Product

Let us examine the uncertainty of a product

$$Y = X_1 X_2 \ ... X_n \tag{5.13}$$

of n independent random variables X_i, each characterised by a probability distribution $p_i(X_i)$. Even though there may be correlations between impacts calculated along parallel pathways (e.g. inhalation and food intake), for the multiplicative processes along a single pathway, independence of the factors is usually a valid assumption. For example, atmospheric dispersion and dose-response function are clearly independent.

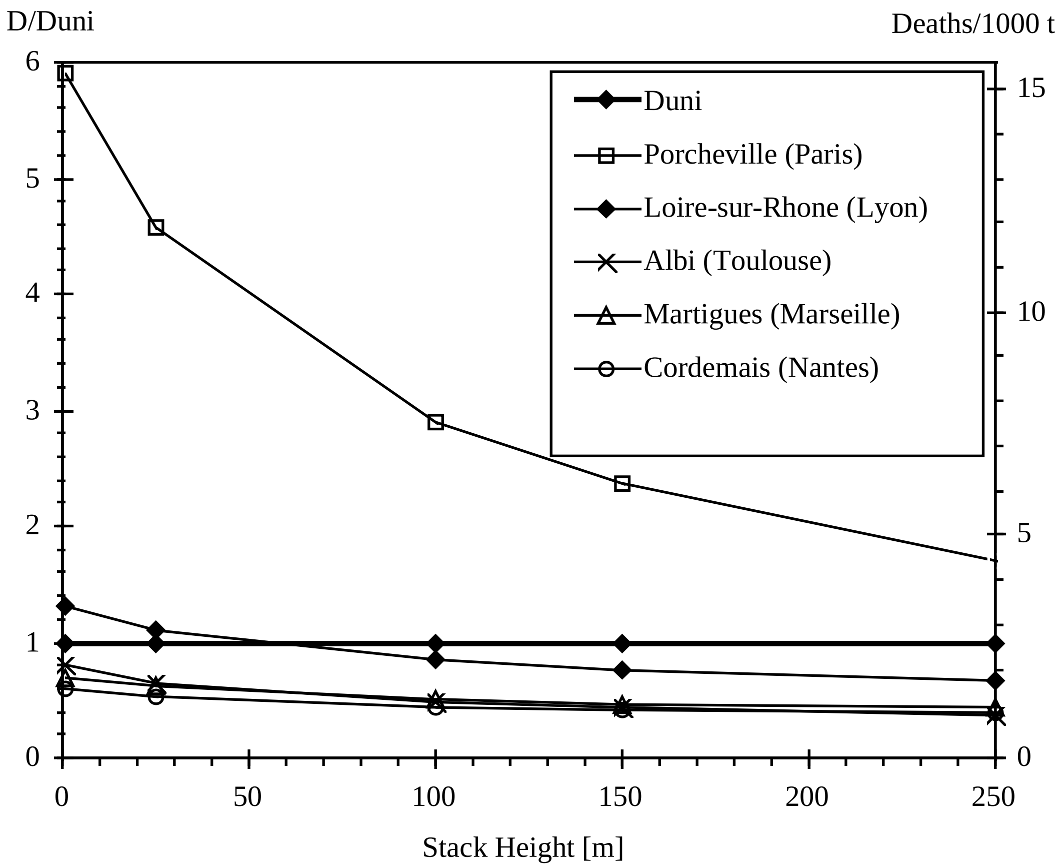

Figure 5.2. Dependence on site and on height of source for a primary pollutant: damage D from SO_2 emissions with linear dose-response function, for five sites in France, in units of

D_{uni} for uniform world model Equation 5.10 (the nearest big city, 25 to 50 km away, is indicated in parentheses). The scale on the right indicates deaths/yr (acute mortality) from a plant with emission 1000 tonnes/yr. Plume rise for typical incinerator conditions is accounted for.

Taking the logarithm of Equation 5.13 we have

$$\ln(Y) = \ln(X_1) + \ln(X_2) + ... + \ln(X_n) \tag{5.14}$$

Let us define the geometric mean μ_{gi} for each variable as the expectation value of the logarithm

$$\ln(\mu_{gi}) = \int p_i(X_i) \ln(X_i)\, dX_i \tag{5.15}$$

for the product we can define in analogous manner

$$\ln(\mu_g) = \int p(Y) \ln(Y)\, dY \tag{5.16}$$

Assuming independence one finds

$$\mu_g = \mu_{g1}\, \mu_{g2} \, ... \, \mu_{gn} \tag{5.17}$$

It will be convenient to work with dimensionless quantities by dividing each variable by its geometric mean

$$x_i = X_i/\mu_{gi} \qquad \text{and } y = Y/\mu_g \tag{5.18}$$

Let us now define the geometric standard deviation σ_{gi} as

$$[\ln(\sigma_{gi})]^2 = \int p_i(X_i) \{[\ln(X_i) - \ln(\mu_{gi})]^2]\}\, dX_i \tag{5.19}$$

Invoking again independence of the distributions, we find that the geometric standard deviation σ_g of the product y is given by

$$[\ln(\sigma_g)]^2 = [\ln(\sigma_{g1})]^2 + [\ln(\sigma_{g2})]^2 + ... + [\ln(\sigma_{gn})]^2 \tag{5.20}$$

This is the key equation on which we will base our uncertainty analysis. It is valid for any set of independent distributions with finite geometric means and geometric standard deviations. The interpretation of the geometric standard deviation in terms of a confidence interval requires an assumption on the probability distribution. For this purpose is instructive to look at the lognormal distribution.

5.4.2 The Lognormal Distribution

The lognormal distribution of a variable x is obtained by assuming that the logarithm of x has a normal distribution (see e.g. Morgan and Henrion 1990). The central limit theorem of statistics, applied to Equation 5.14, implies that the lognormal distribution is the "natural" distribution for multiplicative processes, the same way that the normal (Gaussian) distribution is natural for additive processes. Although the lognormal distribution becomes rigorous only in the limit of infinitely many factors, in practice it can be a good approximation even when the number of factors is quite small (five to ten), provided the spread is not dominated by a distribution that is far from lognormal. For many environmental impacts the lognormal model for the result seems quite relevant because the distributions of the individual factors are not too far from lognormality.

Let U be a Gaussian (or normal) variable with mean ξ and standard deviation ϕ. Its probability density distribution is given by

$$g(u) = \frac{1}{\phi\sqrt{2\pi}} \exp\left(-\frac{(u-\xi)^2}{2\phi^2}\right) \tag{5.21}$$

normalised to unity when integrated over u from $-\infty$ to $+\infty$. Making the change of variable u = ln x we arrive at

$$\int_0^{+\infty} \frac{g(\ln x)}{x} \, dx = 1 \tag{5.22}$$

which leads us to interpret the function

$$f(x) = \frac{g(\ln x)}{x} = \frac{1}{\phi x\sqrt{2\pi}} \exp\left(-\frac{(\ln x-\xi)^2}{2\phi^2}\right) \tag{5.23}$$

as the probability density of a new distribution between 0 and $+\infty$, the lognormal distribution. One can show that the ordinary mean μ and standard deviation σ of the lognormal variable x are given by

$$\mu = \langle x \rangle = \exp\left(\xi + \frac{\phi^2}{2}\right) \tag{5.24}$$

and

$$\sigma^2 = \langle (x - \langle x \rangle)^2 \rangle = [\exp(\phi^2) - 1] \exp(2\xi + \phi^2) \tag{5.25}$$

However, for a lognormal variable the geometric mean μ_g and the geometric standard deviation σ_g are more practical because they provide multiplicative confidence intervals such as

$[\mu_g/\sigma_g, \mu_g.\sigma_g]$ for approximately 68%

and

$[\mu_g/\sigma_g^2, \mu_g.\sigma_g^2]$ for approximately 95% .

The geometric means and standard deviations μ_g and σ_g have been defined by Equations 5.15 and 5.19, and they are related to the parameters ξ and ϕ by

$$\mu_g = \exp(\xi) \quad \text{and} \quad \sigma_g = \exp(\phi) \tag{5.26}$$

The mode x_{max} (where f reaches its maximum) is given by

$$x_{max} = \exp(\xi - \phi^2) \tag{5.27}$$

It is easy to see that $x_{max} \leq \mu_g \leq \mu$, in other words, the mean is always greater than the geometric mean, which itself is greater than the mode. An example of a lognormal distribution is plotted in 5.3. In the limit $\sigma_g \to 1$ it approaches the ordinary normal distribution.

5.5 Sources of Uncertainty

5.5.1 Categories of Uncertainty

In this section we discuss the main contributions to the uncertainty. It is appropriate to group them in four qualitatively different categories:

- technical/scientific

e.g. dose-response functions

- policy/ethical choice,

e.g. value of statistical life and value of a YOLL (year of life lost)

- scenarios for the future

e.g. crop losses may be reduced by the development of more resistant species

- idiosyncrasies of the analyst

e.g. interpretation of ambiguous information, or human error.

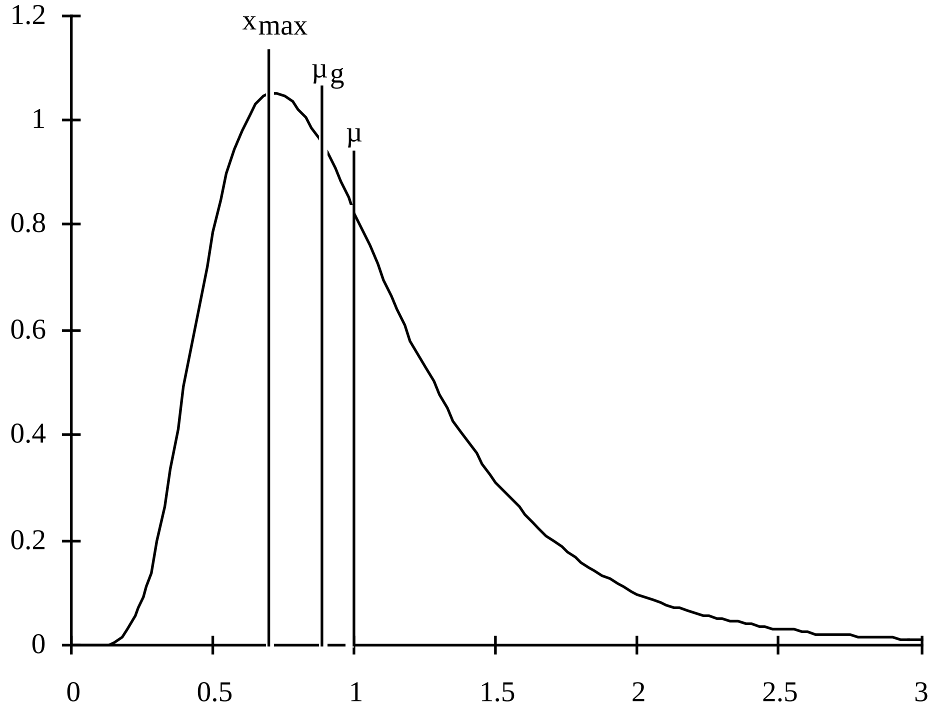

Figure 5.3. Example of a lognormal probability density distribution, with mean $\mu = 1$, standard deviation $\sigma = 0.5$, geometric mean $\mu_g = 0.89$, geometric standard deviation $\sigma_g = 1.6$ and mode $x_{max} = 0.72$.

The best way of dealing with the second and third of these categories is to indicate how the results depend on these choices and present numbers for different scenarios if the effect on the result is not obvious. One needs to guard against overconfidence - a danger highlighted by several interesting papers (Henrion and Fischhoff 1985, Shlyakhter 1994). The latter has developed an interesting model for quantifying unsuspected uncertainties.

5.6 Quantifying Technical/Scientific Uncertainty

5.6.1 Emissions

General engineering considerations suggest that uncertainties in the emissions of the major pollutants from combustion (per time of operation or per unit of energy) are relatively small for installations that are subject to monitoring, perhaps on the order of a few percent for CO_2 and about ten to twenty percent for PM_{10}, SO_2 and NO. For the calculations we assume a geometric standard deviation $\boldsymbol{\sigma_g}$ **of 1.2**; but note that in any case it hardly matters because the difference between 1 and 1.2 is entirely negligible compared to the other contributions to σ_{gtot}. By contrast, emissions from various other activities, e.g., methane leaks from gas pipelines, are poorly known; they could easily be off by a factor of two.

Emissions of trace pollutants, such as lead and mercury from combustion of coal or wastes, may also be very uncertain because they vary with the composition of the fuel.

5.6.2 Dispersion Models

A geometric standard deviation in the range from three to five is sometimes cited for the uncertainty of atmospheric dispersion models (Cohrssen and Covello 1989), but without making a distinction between episodic values and averages over space or time. Atmospheric models are most accurate for annual average and for pollutants without chemical transformation (a fair approximation for particulates and for SO_2, up to distances of several hundred km). The models are less reliable when they involve chemical reactions in addition to transport, e.g. the formation of ozone. The most uncertain calculations are those for specific episodes. As an interesting example one can cite the international competition to test models for the dispersion of the fallout from Chernobyl (Klug et al. 1993). Here the goal was to predict how much fallout was received at various times and places of the earth; this was a very ambitious goal and the discrepancies between predictions and data turned out to span several orders of magnitude.

As implied by the analysis in Section 5.4; for total damage the uncertainty of a dispersion model is determined by the uncertainty in the removal velocity as defined by Equation 5.8. For an indication of the kind of distribution that can be expected, we show, in Figure 5.4, a histogram for data for dry deposition velocity of SO_2, based on a review by Sehmel (1980). Visibly, a logarithmic scale is much more appropriate for these data than a linear one. The geometric standard deviation is approximately 2.5 for this sample. The variability of this sample reflects two distinct effects: variability due to different surface materials, and uncertainty for a given surface material. In practical applications uncertainties in the geographic distribution of surface materials blur the distinction. More recent data would probably have a smaller standard deviation, but we have not been able to find a survey as comprehensive as that of Sehmel. For particulates we refer to Figure 16.20 of Seinfeld (1986), which likewise suggests a lognormal distribution.

The possibility of low values of dry deposition velocities could imply very large damages under dry conditions. However, for the wet climates typical of Europe, long distance dispersion will be limited by wet deposition. This has been verified for particulate matter by varying the dry deposition velocity in EcoSense Thus the uncertainty of total deposition appears to be significantly smaller than suggested by dry deposition data.

Thus we are led to suggest a **σ_g around 2** for dispersion modelling of primary air pollutants, as far as total damages are concerned. These numbers are consistent with estimates by McKone and Ryan (1989).

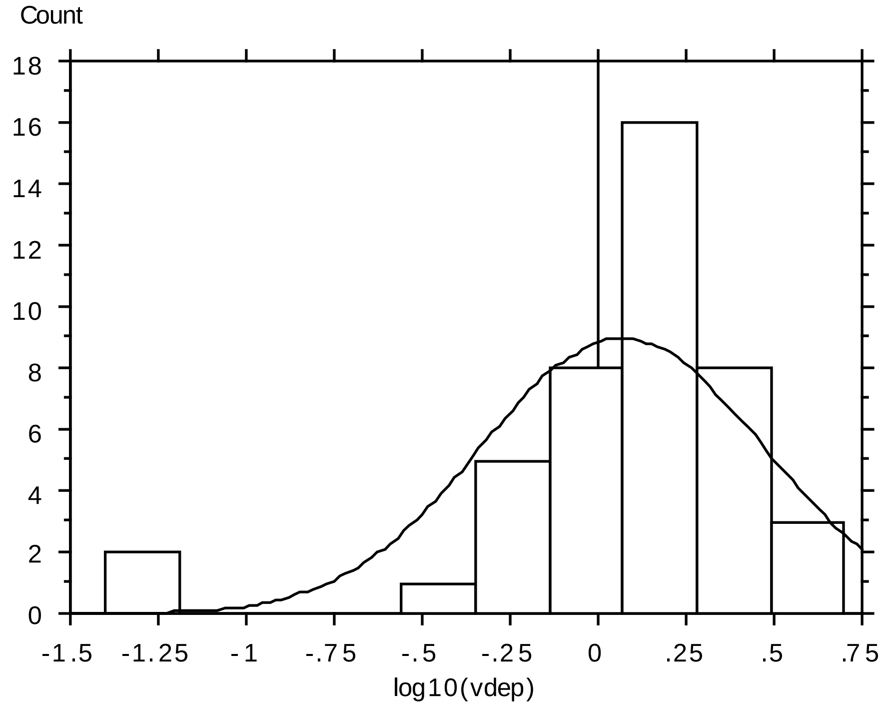

Figure 5.4. Distribution and lognormal fit of maximum values, in the review of Sehmel (1980), for dry deposition velocity (in cm/s) of SO_2 over different surfaces.

5.6.3 Dose-Response Functions

The uncertainty of dose-response functions varies widely from case to case. Best established are the ones for health impacts from radionuclides, the ones for certain health impacts from the major pollutants (PM_{10}, SO_2, NO_2 and O_3), and the ones for impacts of SO_2, NO_2 and O_3 on certain crops whose economic importance has prompted laboratory studies. But extrapolation to other species is problematic. Comparing dose-response functions for different species, one finds that the sensitivities to air pollution can easily vary by an order of magnitude from one species to another. However, for the classical air pollutants this does not matter since these exposure-response functions have been determined by epidemiology.

The confidence intervals of exposure-response functions are usually reported for 95% probability, and they are approximately symmetric (of the form $\mu\pm\delta$) around the mean μ. The underlying probability distributions (implicit in the regression software used in the respective studies) are usually not lognormal, hence it is necessary to estimate the corresponding geometric standard deviations σ_g by an approximation. For this purpose we suppose that $\pm\delta/2$ corresponds to a 68% confidence interval, as for a Gaussian distribution. Then we fit the corresponding lognormal distribution such that its 68% confidence interval equals $[\mu-\delta/2, \mu+\delta/2]$, which yields σ_g as

$$\sigma_g = \sqrt{\frac{\mu + \delta/2}{\mu - \delta/2}} \qquad (5.28)$$

Typically one finds **σ_g in the range 1.2 to 1.8**. For the chronic mortality study of Pope et al one finds **1.3**.

For a YOLL calculation of mortality there are additional factors in the calculation of costs. For acute mortality the cost is proportional to the life span reduction ΔT. There are no data for that; the exposure-response function reports only a change in the number of deaths per day, independent of ΔT. To estimate the corresponding σ_g, let us examine plausible upper and lower limits for the population-averaged ΔT due to acute mortality. The lower limit is at least a few days, otherwise the effect could not be observed by time series studies. The upper limit seems unlikely to be more than a few years. Taking 4 days and 1024 days as 95% confidence interval, one obtains a **σ_g of 4** if the distribution is lognormal (the corresponding median ΔT is 64 days).

For chronic mortality one needs to determine the YOLL from change in the age-specific mortality rate reported by these studies. This step is uncertain because we have no data on the duration of relevant exposure and on the time delay between exposure and death. Based on various calculations performed by members of the ExternE team (Donnan and Hurley, Krewitt, Markandya, Rabl), we estimate that the corresponding uncertainty might be a **σ_g of 2**.

There is, however, another type of uncertainty that can be highlighted by writing the dose-response function in the form

damage = f(dose, **other ?**) (5.29)

where "other ?" indicates all the variables that have not been taken into account. Transferring a dose response function to a situation different from the one for which it has been derived is problematic if there is a difference in a variable that has not been taken into account. For example, the dose-response functions for particulate matter give no indication of the composition of the particles. If the particulates (typically reported as TSP) from the power plant under study are different from the ones on which the function has been based (typically PM_{10} or $PM_{2.5}$), a correction factor must be estimated.

This issue of the composition of particulates is particularly troubling for the transfer of exposure-response functions from the USA to Europe. Presumably the biology of the populations is not very different, including age distribution and the practice of medicine. On the other hand, the composition of the particles in the air is likely to be quite different; in particular the percentage of secondary nitrate and sulphates aerosols. One indication for that can be seen from the emissions data in Table 5.1. On average the ratio of nitrates (sulphates)

to primary particulates is proportional to the ratio of NO_2 (SO_2) and TSP emissions. Table 5.1 shows that the nitrate ratio is about twice as high in Europe as in the USA.
If nitrates are harmless, the toxicity of ambient particulate matter in the USA could be higher than in Europe. Could this explain why PM_{10} exposure-response functions for the same end points have about twice as high a slope in the USA as in Europe? For the calculations of σ_{gtot} we estimate that the corresponding uncertainty might be a **σ_g of 2**.

Effects not taken into account may make the real uncertainty even larger. Usually the dose-response function is the weakest link in the impact pathway chain, and its uncertainty is much larger than the confidence interval reported in the respective studies because the latter accounts only for random errors in a particular data set, not systematic errors.

Table 5.1 Emissions data for 1990, from OECD Environmental Data Compendium (1995).

1000 t/yr	**TSP**	**NO_2**	**NO_2/TSP**	**SO_2**	**SO_2/TSP**
USA	7345	21373	2.9	19518	2.7
France	234	1487	6.4	1200	5.1
West Germany	436	2460	5.6	878	2.0
UK	460	2731	5.9	3754	8.2

5.6.4 Economic Valuation

Some physical impacts can be easily valued by their price on the market, e.g. the price of crops. There is little uncertainty of these prices as quoted at any particular place and time; uncertainty comes mainly from their variability and from possible errors in collecting the information. Geometric standard deviations around 1.1 seem reasonable. Non-market goods are difficult to value economically. This is especially true for the reference value for the protection of human lives, often called value of statistical life (VOSL). It involves an ethical choice, and now there is an emerging consensus in democratic countries that one should base it on individual preferences rather than human capital. It seems to be the most difficult good to monetise, and the uncertainty is large. The distribution of VOSL results from various studies of individual preferences tends to be lognormal, as illustrated for example by Figure 5.5 which is based on the Ives, Kemp and Thieme (1993) survey of 78 VOSL studies published between 1973 and 1990. Figure 5.5 gives equal weight to all studies, regardless of quality or age; the resulting large spread of values could probably be reduced by applying reasonable selection criteria based on the benefit of hindsight.

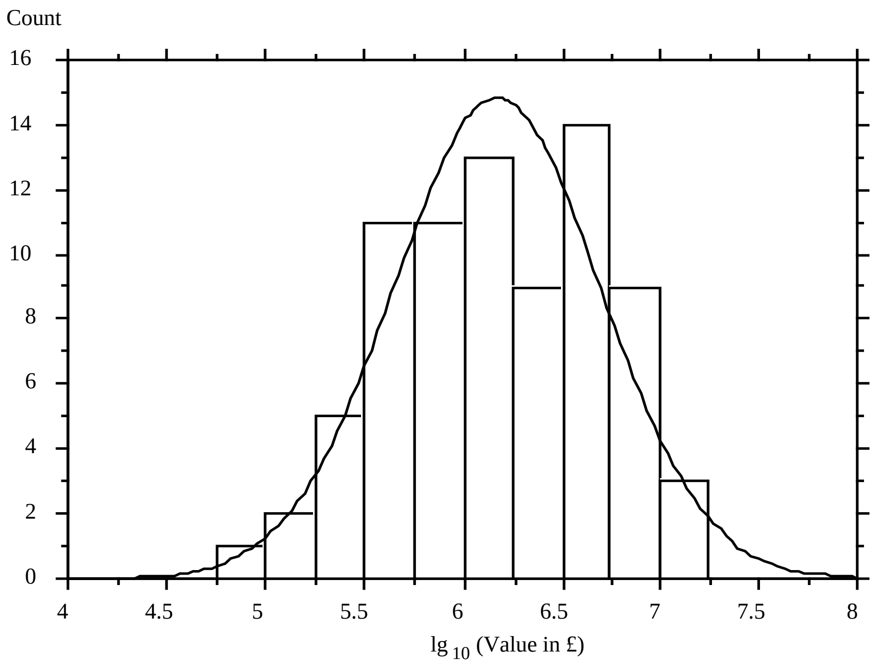

Figure 5.5 Example of lognormal distribution for economic valuation: reference value for protection of human life, in £1990, as determined by 78 studies reviewed by Ives et al. (1993).

The lognormal distribution is asymmetrical, and it can have a large tail of high outliers. As a consequence, if the spread is large, the mean is much larger than the median. This is illustrated in Table 5.2 which summarises the distributional characteristics of Ives, Kemp and Thieme (1993) as well as of another survey, carried out by Markandya (EC 1995c) and limited to European studies. The spread is so large, with a geometric standard deviation around 3.4, that even a one-standard deviation interval extends to negative values. Clearly it does not make much sense to use ordinary mean and standard deviation in such cases. The median is far less affected by outliers, and in Table 5.2 it is fairly close to the geometric mean. These considerations are implicit in the weighting of the 15 studies in EC (1995c) that led to a recommended value of 2.60 Million ECU ($ 3.2 Million) - close to the median and between median and geometric mean.

Table 5.2. Uncertainty of the value of statistical life: distributional characteristics from two surveys.

	Ives, Kemp and Thieme (1993)	EC (1995c)
	78 studies, world wide 1973-1990 in Million £ (1 £1990 = 1.78 $)	15 studies, Europe 1973-1990 in Million ECU (1 ECU = 1.24 $)
mean	2.76	3.93
standard deviation	3.00	4.21
median	1.59	2.70
geometric mean μ_g	**1.49**	**2.16**
geometric standard deviation σ_g	**3.35**	**3.41**

Table 5.3 Exposure-response functions and costs for "core" and "sensitivity" for PM_{10}, the latter in *italics*. Exposure-response functions are expressed in units of cases per average population.

End point and Reference	**f_{E-R}** cases/(pers.yr.µg/m^3)	**Cost** ECU/case	**Cost** ECU/(pers.yr.µg/m^3)	**%**
ASTHMATICS adults Bronchodilator usage (Dusseldorp et al. 95)	4.56E-03	37	1.69E-01	0.31%
ASTHMATICS adults (CoughDusseldorp et al.95)	4.69E-03	7	3.28E-02	0.06%
ASTHMATICS adults Lower respiratory symptoms (Dusseldorp et al. 95)	1.70E-03	7.5	1.27E-02	0.02%
ASTHMATICS children Bronchodilator usage (Roemer et al. 93)	5.43E-04	37	2.01E-02	0.04%
ASTHMATICS children Cough (Pope/Dockery 92)	9.34E-04	7	6.54E-03	0.01%
ASTHMATICS children Lower respiratory symptoms (Roemer et al. 93)	7.20E-04	7.5	5.40E-03	0.01%
ASTHMATICS ELDERLY 65+ Congestive heart failure (Schwartz/Morris 95)	2.59E-06	7870	2.04E-02	0.04%
ADULTS Chronic bronchitis (Abbey et al. 95)	3.92E-05	105000	4.12E+00	7.47%
ENTIRE POPULATION Chronic Mortality (CM) (Pope et al. 95)	5.76E-04	84330	4.86E+01	88.10%
ADULTS Restricted activity days (RAD) (Ostro 87)	2.00E-02	75	1.50E+00	2.72%
ASTHMATICS CHILDREN Chronic bronchitis (Dockery et al. 89)	3.22E-04	225	7.25E-02	0.13%
ASTHMATICS CHILDREN Chronic cough (Dockery et al. 89)	4.14E-04	225	9.32E-02	0.17%
ENTIRE POPULATION Cerebrovascular hospital admissions (Wordley et al. 97)	5.04E-06	7870	3.97E-02	0.07%
ENTIRE POPULATION Respiratory hospital admissions (RHA) (Dab et al. 96)	2.07E-06	7870	1.63E-02	0.03%
Total "core"			5.47E+01	99.17%
ERV for COPD, all (Sunyer et al 1993)	*7.20E-06*	*223*	*1.61E-03*	*0.00%*
ERV for asthma, all (Schwartz93/Bates90)	*6.45E-06*	*223*	*1.44E-03*	*0.00%*
ERV for croup, all (Schwartz et al 91)	*2.91E-05*	*223*	*6.49E-03*	*0.01%*
Symptom days, all (Hurley)	*5.71E-02*	*7.5*	*4.28E-01*	*0.78%*
Isch heart dis, >65 (Schwartz&Morris95)	*2.45E-06*	*7870*	*1.93E-02*	*0.03%*
Total "sensitivity"			*4.57E-01*	0.83%
Total all			**5.51E+01**	100%

Table 5.4 shows the analogous results for CO. Here, by contrast to PM_{10}, the "sensitivity" functions increase the damage by almost a factor of 30. However, since the CO damage is an extremely small part of the total, the effect of these functions on the total damage is small.

Table 5.4 Exposure-response functions and costs for "core" and "sensitivity" for CO, the latter in *italics*. Exposure-response functions are expressed in units of cases per average population.

End point and Reference	$f_{E\text{-}R}$ cases/(pers.yr.µg/m^3)	Cost ECU/case	Cost ECU/(pers.yr.µg/m^3)	%
ASTHMATICS all ELDERLY 65+ Congestive heart failure (Schwartz/Morris, 95)	7.90E-08	7870	6.21E-04	3.5%
Isch heart dis, >65 (Schwartz&Morris95)	*5.84E-08*	*7870*	*4.59E-04*	*2.6%*
Actue YOLL, all (Touluomi et al 94)	*1.08E-07*	*155000*	*1.67E-02*	*93.9%*
Total CO, includ. "sensitivity"			**1.78E-02**	100.0%

The role of the "sensitivity" functions for NO_x is more complicated because direct effects of NO_x have not been included so far, and thus their inclusion in the calculation is site dependent. However, a simple calculation for an extreme example shows that the "sensitivity" functions for NO_x can also be neglected. As an extreme example we take a trip by car in a very large city (between the two airports of Paris, Roissy and Orly); this provides an upper bound on the importance of direct NO_x effects because nowhere else is there such a large local population that is affected by a primary pollutant. For rural sites the damage ratio primary/secondary pollutant is about an order of magnitude smaller.

Table 5.5 Effect of "sensitivity" exposure-response functions for NO_x (in *italics*), for trip by car in Paris.

	Diesel car		Gasoline car, 3-way catalyst	
Emissions and Damage	**g/km**	**mECU/km**	**g/km**	**mECU/km**
$PM_{2.5}$ (mortality)	1.74E-01	476	1.73E-02	47.6
NO_X "sensitivity" E-R functions	*7.48E-01*	*9.3*	*6.81E-01*	*8.5*
NO_X damage via XNO_3 (mortality)		16.2		14.4
NO_X damage via O_3		1.1		1.0
Total (all impacts)		562		76
Comparisons				
NO_X "sensitivity"/$PM_{2.5}$		0.02		0.18
NO_X "sensitivity"/XNO_3		0.59		0.59
NO_X "sensitivity"/O_3		8.4		8.4
NO_X "sensitivity"/Total		0.017		0.11

The results are shown in Table 5.5, in a format that highlights the comparison of the direct NO_x effect with other impacts. As a fraction of the total (health, crops, buildings etc.) the "sensitivity" functions for NO_x add less than 2% for the diesel and 11% for the gasoline car. The direct NO_x effect is also small compared to the $PM_{2.5}$ impacts that are considered relatively certain.

We conclude that none of the "sensitivity" functions have a significant effect on the total damage. For particulates and aerosols the sensitivity functions increase the "core" estimates by less than one percent, independent of site. For CO the sensitivity functions increase the "core" estimates by about a factor of 30, also independent of site; however, the effect on total damage is small in any case. Only for NO_x does the effect of the sensitivity functions depend on the emission site. But even for the extreme case of motor vehicle emissions in Paris the effect on the total damage is small; for most other sites it is negligible.

Some additional sensitivity tests have been suggested:

(i) drop the direct SO_2 impacts. This is analogous, in reverse, to the case of the direct NO_x impacts discussed above. As in the case of nitrates, the effect is small.

(ii) treat all particles as PM_{10} or $PM_{2.5}$. This involves a straightforward multiplication of the respective PM results by 0.60 or 1/0.6. Since the dominant damage costs are due to PM, the effect is easy to see as a corresponding increase in the total cost, especially for cases where a single particulate type (primary particles, nitrates or sulphates) dominates the total cost.

(iii) omit the RADs. The effect is small since RADs contribute less than 3% of the PM cost.

(iv) Scale down the chronic mortality of Pope et al by a factor of 2. This roughly divides the cost by two since chronic mortality is the predominant cost.

5.8 Sample Results and Conclusions

We have shown that the impact pathway analysis is essentially multiplicative, when one sums over all receptor sites to obtain the total damage caused by a pollutant. If the uncertainty of each of the steps of the pathway analysis is characterised by a geometric standard deviation, the geometric standard deviation of the result is readily obtained by Equation 5.20. It involves a quadratic sum of the terms of the pathway steps, and it is dominated by the largest errors. The central limit theorem implies that the distribution of the result is likely to be approximately lognormal; thus it is determined by its geometric mean μ_g and geometric standard deviation σ_g. These have a natural interpretation in terms of multiplicative confidence intervals about the median (which is approximately equal to the geometric mean).

We have discussed sources of uncertainty and provided estimates of geometric standard deviations for each. We have examined data for uncertainty distributions for two particularly important parameters: the value of life, and the deposition velocity. They seem lognormal. We also have examined special complications that can arise when dose-response function is not positive definite.

Using these component uncertainties for the steps of the pathway, we illustrate the method by estimating the uncertainty of acute mortality from particulate air pollution; this is a dominant impact for coal fired power plants (ORNL/RFF 1994, EC 1995, Rowe et al 1995, Curtiss et al 1995). The details are shown in Table 5.6 The geometric standard deviations for removal velocity is perhaps too large, being based on data published fifteen years ago (Sehmel 1980).

From the results of this Chapter we recommend a label A for hospital admissions, a label B for Chronic mortality and a label C for acute mortality. Labels for other endpoints can be found in Tables 5.7 and 5.8.

Table 5.6 Sample calculations of geometric standard deviation σ_{gtot} for health costs due to particulates, using Equation 5.20 and the component uncertainties as identified in Section 5. Interpretation in terms of approximate multiplicative confidence intervals about the geometric mean μ_g [≈ median), e.g., [μ_g/σ_g, $\mu_g.\sigma_g$] for 68% and [μ_g/σ_g^2, $\mu_g.\sigma_g^2$] for 95% confidence.

PM_{10}	Chronic Mortality	Acute Mortality	Hospitalisation
Emission, TSP	1.2	1.2	1.2
Dispersion	2	2	2
f_{E-R} regression	1.3	1.3	1.3
f_{E-R} transfer (TSP∅PMx, composition)	2	2	2
Cost per day			1.2
Duration			1.2
YOLL (years of life lost)	1.5	4	
VOSL (value of statistical life)	2	2	
Value of YOLL (including discount rate)	1.3		
Latency (including discount rate)	1.4		
Total	**4.0**	**6.4**	**2.9**
Without VOSL	3.2	5.6	

Table 5.7 Uncertainty labels for human health impacts for the *ExternE Core Project*.

Receptor	Impact Category	Reference	Pollutant	Uncertainty Label
Asthmatics				
Adults	Bronchodilator usage	Dusseldorp et al., 95	PM_{10}	B
	Cough	Dusseldorp et al., 95	PM_{10}	A
	Lower respiratory symptoms (wheeze)	Dusseldorp et al., 95	PM_{10}	A
Children	Bronchodilator usage	Roemer et al., 93	PM_{10}	B
	Cough	Pope/Dockery, 92	PM_{10}	A
	Lower respiratory symptoms (wheeze)	Roemer et al., 93	PM_{10}	A
All	Asthma attacks	Whittemore/Korn, 80	O_3	B?
Elderly 65+				
	Congestive heart failure	Schwartz/Morris, 95	PM_{10} CO	B B
Children				
	Chronic bronchitis	Dockery et al., 89	PM_{10}	B
	Chronic cough	Dockery et al., 89	PM_{10}	B
Adults				
	Restricted activity days Minor restricted activity day	Ostro, 87 Ostro/Rothschild, 89	PM_{10} O_3	B B
	Chronic bronchitis	Abbey et al., 95	PM_{10}	A
All				
	Respiratory hospital admissions	Dab et al., 96 Ponce de Leon, 96	PM_{10} SO_2 O_3	A A A
	Cerebrovascular hospital admissions	Wordley et al., 97	PM_{10}	B
	Symptom days	Krupnick et al., 90	O_3	A
	Cancer risk estimates	Pilkington et al., 97	Benzene Benzo[a]Pyrene 1,3 butadiene Diesel particles	A A A A
	Acute Mortality	Koln/Amsterdam, 96 Sunyer et al., 96 London/Athens, 96 Sunyer et al., 96	PM_{10} SO_2 O_2	C C C
	Chronic Mortality	Pope et al., 95	PM_{10}	B

Table 5.8 Human health exposure-response functions for *sensitivity analysis only* (Western Europe).

Receptor	Impact Category	Reference	Pollutant	Uncertainty Label
Elderly, 65+				
	Ischaemic heart disease	Schwartz/Morris, 95	PM_{10} CO	B B
All				
	Respiratory hospital admissions	Ponce de Leon, 96	NO_2	A?
	ERV for COPD	Sunyer et al., 93	PM_{10}	B
	ERV for asthma	Schwartz 93/Bates 90 Cody 92 / Bates 90	PM_{10} O_3	B B
	ERV for croup in pre-school children	Schwartz et al., 91	PM_{10}	B
	Cancer risk estimates	Pilkington et al., 97	Formaldehyde	B?
	Acute Mortality	Touluomi et al, 94 Barcelona/London, 96	CO NO_2	C? C?

ERV = emergency room visits
COPD = chronic obstructive pulmonary disease

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6. ASSESSMENT OF GLOBAL WARMING DAMAGES

6.1 Introduction

In the first stages of the ExternE Project (European Commission, 1995) global warming estimates were largely based on three studies (Cline, 1992; Fankhauser, 1993; Tol, 1993). The 1995 IPCC Working Group III report (Bruce et al, 1996) reviewed these and other studies and reported from them a range of damages from \$5 to \$125 per tonne of carbon emitted in 1995. However, the IPCC stated that this range did not fully characterise uncertainties, leaving them unable to endorse any particular figure or range.

Much previous work concentrated on quantifying damages at the point in time when the CO_2 level is doubled against that which prevailed in 'pre-industrial times', paying little attention to damages at other levels of climate change or the rate of climate change. It seems reasonable to postulate that effects would be lower if climate change happens slowly than if it happens quickly. This would give people a longer time to react and take mitigating actions, such as changing to new crop types, planning orderly evacuation of places that face an increasingly unacceptable risk of catastrophic flooding, and so on. It is thus important to take account of different scenarios, and to follow them over time, rather than basing estimates on a single point in the future.

In 1992 the IPCC proposed a set of 6 scenarios, or 'possible futures'. They extend to the year 2100, and differ with respect to a number of factors, including;

- population
- GDP growth
- total energy use
- use of specific energy sources (nuclear, fossil, and renewable)

Given the uncertainties involved in making any statement about the future, no judgement was given by IPCC as to which scenario(s) appeared most likely. Although these scenarios do not provide all of the socio-economic information needed to assess damages they do provide a good baseline for comparable damage assessment. Until now, however, they have not been well integrated into damage assessment work.

From consideration of numerous issues it was concluded that continued reliance on estimates of global warming damages from other studies was no longer acceptable. Within the phase of ExternE reported here that a careful examination of the issues was made, to look further at the uncertainties that exist in the assessment. This demonstrated the analytical problems of the impact assessment, arising from there being a very large number of possible impacts of climate change most of which will be far reaching in space and time.

It also demonstrated the problems of valuation of these impacts, in which difficult, and essentially normative, judgements are made about:

- discount rate;
- the treatment of equity;
- socio-economic conditions;
- climate and impact uncertainties;
- treatment of sustainability problems.

These issues have now been explored in more depth using two models - FUND, developed by Richard Tol of the Institute for Environmental Studies at the Vrije Universiteit in Amsterdam, and the Open Framework, developed by Tom Downing and colleagues at the Environmental Change Unit at the University of Oxford. So far as is reasonable, the assumptions within the FUND and Open Framework models are both explicit and consistent. The models have some common features, both calculating greenhouse gas marginal damages. However, the models are very different in structure and purpose, so that convergence is neither possible nor desirable. Comparing results to improve understanding of the issues has therefore been the goal of the exercise, not model convergence Another major advantage over previous work is that the models both enable specific account to be taken of the scenarios developed by IPCC.

Numerous impacts are included in the two models, ranging from effects on agricultural production to effects on energy demand. Details of precisely what is included and excluded by the two models are provided by European Commission (1998), from which this Chapter is abstracted. The repetition within the series of reports is clearly warranted given the importance of global warming in fuel cycle assessment.

6.2 Interpretation of Results

Section 6.4 of this chapter contains selected results for the base case and some sensitivity analyses. The results given have been selected to provide illustration of the issues that affect the analysis - they are not a complete report of the output of the ExternE global warming task team.

Like the range given by IPCC, the ranges given here cannot be considered to represent a full appraisal of uncertainty. Only a small number of uncertainties are addressed in the sensitivity analysis, though it seems likely that those selected are among the most important. Even then, not all the sensitivities are considered simultaneously. Monte-Carlo analysis has been used with the FUND model to describe confidence limits. However, this does not include parameters such as discount rate that are dealt with in the sensitivity analysis. The IPCC conclusion, that the range of damage estimates in the published literature does not fully characterise uncertainties, is thus equally valid for these new estimates.

In view of these problems, and in the interests of providing policy makers with good guidance, the task team has sought (though inevitably within limits) to avoid introducing personal bias on issues like discount rate, which could force policy in a particular direction.

There is a need for other users of the results, such as energy systems modellers or policy makers, to both understand and pass on information regarding uncertainty, and not to ignore it because of the problems that inevitably arise. **The Project team feels so strongly about this that reference should not be made to the ExternE Project results unless reference is also made to the uncertainties inherent in any analysis and our attempts to address them.** Reliance on any single number in a policy-related context will provide answers that are considerably less robust than results based on the range, although this, in itself, is uncertain.

6.3 Key Issues

Some key issues have been identified which potentially have very important effects on the assessment of climate change damages - equity, discounting, socio-economic conditions, climate and impact uncertainties, and the treatment of sustainability problems. These have been reviewed. In some cases no approach can be recommended as uniquely correct, and instead we adopt a "base case" against which to measure sensitivities.

6.3.1 Discounting Damages Over Protracted Timescales

The task team reported results for different discount rates (see below, and European Commission, 1998). At the present time the team do not consider it appropriate to state that any particular rate is 'correct' (for long term damages in particular this is as much a political question as a scientific one), though the task team tended towards a rate of the order of 1 or 3% - somewhat lower than the 5% that has been used in many other climate change damage analyses. This stresses the judgmental nature of some important parts of the analysis. However, it also creates difficulty in reporting the results and identifying a base case, so is worthy of additional consideration. The figure of 3% was originally selected as the base case *elsewhere* in ExternE from the perspective of incorporating a sustainable rate of per capita growth with an acceptable rate of time preference.

However, it has subsequently been argued that, for Inter-generational damages[9], individual time preference is irrelevant, and therefore a discount rate equal to the per capita growth rate is appropriate (see Rabl, 1996). In the IPCC scenarios the per capita growth rate is between 1% and 3%, but closer to the former. If this line of argument is adopted, a 1% base case is preferable though there are theoretical arguments against it. A rate of 3% seems theoretically more robust, but has more significant implications for sustainability. The literature on climate change damage assessment does not provide clear guidance (with rates ranging up to 5%).

It is therefore necessary to look in more detail at the consequences of using different discount rates for analysis of damages that occur in the long term future (Figure 6.1). A rate of 10% (typical of that used in commercial decision making) leads after only 25 years to damages falling to a negligible level (taken here for illustration as being less than 10% of the original damages).

[9] Intergenerational damages are those caused by the actions (e.g. greenhouse gas emissions) of one generation that affect another generation.

For a 3% discount rate this point is reached after 77 years. For 1% it is reached after 230 years. The use of a rate of 10% clearly looks inappropriate from the perspective of soft-sustainability to which the European Union is committed, given long term growth rates. However, the choice between 3% and 1% on grounds of soft-sustainability is not so clear.

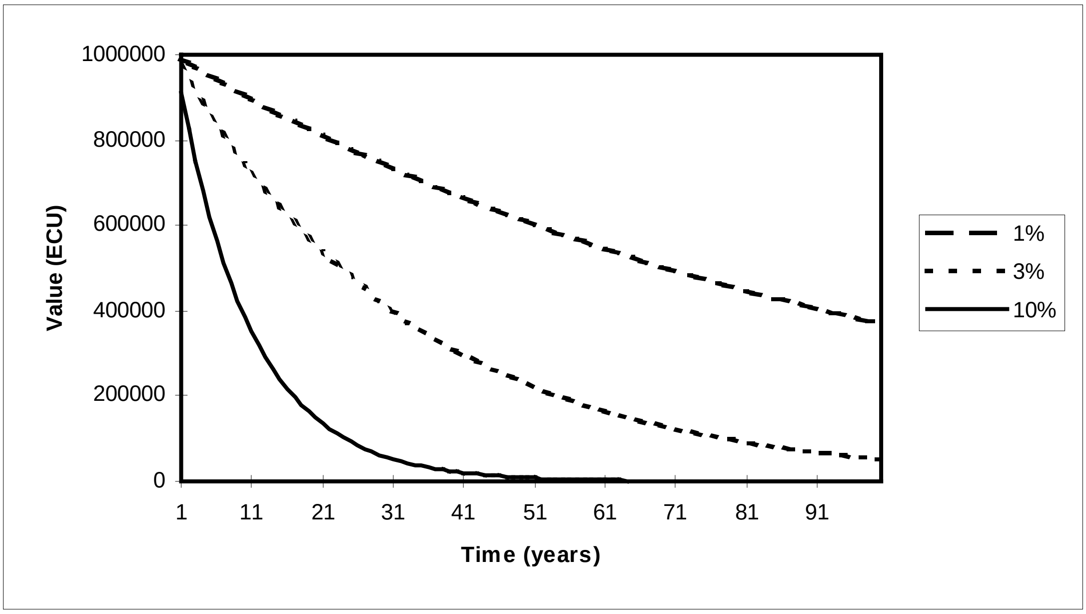

Figure 6.1 Effect of discount rate on present value of damages worth 1 million ECU at the time (from 1 to 100 years in the future) when damage is incurred.

Given the nature of the ExternE project, some consideration of other types of damage is important as a check on consistency. The most extreme example concerns the consequences of long term disposal of high level radioactive waste. These are commonly assessed over periods of 10,000 years or more. The use of any discount rate more than marginally above zero would reduce damages to a point at which they would be considered negligible in a fraction of this time. Even using a rate of 1%, any damage occurring in 10,000 years time would need to be divided by a factor of 1.6×10^{43} to obtain present value. The simple fact that such extended time-spans are considered necessary for assessment of some forms of environmental damage suggests that policy makers do not consider traditional economic analysis to apply in the long term.

Variation of the discount rate over time might seem appropriate, but, at least without assumptions about long term economic performance and the preferences of future generations, there is little information available for this to be done in a way that is any more defensible than the use of a small and constant rate for all Inter-generational effects.

The discount rate clearly has a very large effect on the net present value of damage incurred in the far future, and therefore on the marginal costs of current emissions.

There are some arguments that "pure time preference" is not an admissible factor in assessing inter-generational damages, and therefore that the rate adopted should be equal to the long term per capita growth rate. This would imply scenario dependent discounting which we have not implemented. There is broad agreement that a low positive discount rate should be used, but not its exact value. The full report presents results at a number of discount rates - 0%, 1%, 3% and 10% - but does not recommend the use of the extremes of this set.

6.3.2 The Treatment of Equity

Equity issues have been controversial in the political discussions of climate change damages. "Willingness to pay (WTP)" is the standard measure of value in environmental economics, adopted throughout the ExternE Project. However, WTP is a function of income and therefore lower in poorer countries, so there are equity objections to simple additive aggregation. We reject the approach of using common values for all countries. Instead we prefer to adopt WTP values but to address equity concerns in the aggregation process, using weighting factors to account for declining marginal utility of income.

6.3.3 Socio-economic Conditions

The future damages imposed by climate change due to current emissions will depend on the socio-economic conditions in future generations, in particular the capacity of those societies to adapt to the impacts of climate change. Marginal damages of emissions are therefore dependent on socio-economic as well as climatic scenarios. We have adopted two IPCC scenarios for examination - IS92a and IS92d - which may be thought of as representing "trend projection" and "more sustainable development" respectively. However, both are under-specified for assessing climate change impacts. The additional assumptions required may be critical, especially the regional climate changes and socio-economic development patterns for those societies most vulnerable to impacts of climate change. In these cases, there are potentially some additional damage categories - like famine and conflict - the risk of which will depend largely on the underlying socio-economic conditions, but which climate change could exacerbate. We call these "socially contingent damages" and note that they are difficult to estimate.

6.3.4 Climate and Impact Uncertainties

Uncertainties arise at all stages of the analysis. The sensitivity of global climate change to greenhouse concentrations is still rather uncertain. The pattern of expected regional climate change is even less well established. The climate change impact literature is still in its infancy, especially with respect to adaptive capacity, so that impact assessments are incomplete and have large uncertainties. Accurate damage assessment is not therefore feasible. In the Open Framework model some estimates of uncertainty have been made carrying through upper and lower estimates. A formal uncertainty analysis has been undertaken with the FUND model. Both analyses rely on expert judgement of levels of uncertainty. In addition, some of the most important uncertainties are not statistical, but depend on ethical/political choices (e.g. treatment of equity, discount rate). In these cases we have examined the sensitivity of marginal damages to different choices.

6.3.5 Treatment of Sustainability Problems

The assessment of external costs lies explicitly in the paradigm of weak sustainability (see Chapter 2). For climate change damages, as in the rest of ExternE, it can prove difficult to place marginal values on natural systems, in particular where they are not substitutable and/or the environmental problem is better characterised by scale limits. It can be argued that the concepts of strong sustainability are more applicable in these cases. The potential for developing an intermediate framework based on “safe minimum standards” was explored in co-operation with the ExternE Sustainability Indicators Task (Atkinson *et al*, 1997).

6.4 Key Impacts

Climate change has a very large number of impacts. The literature has been reviewed to identify those impacts which are of most interest to policy-makers and most likely to result in significant damages. We conclude that these are likely to be the impacts of sea level rise and extreme weather events as well as impacts on human health, agriculture, water resources and ecosystems. We have concentrated on these impact categories, as well as some others already included in the FUND and Open Framework models such as energy demand and migration.

Health impacts have been reviewed in some detail. Heat stress and cold stress impacts will be influenced in opposite directions so that the net impact (globally) of direct temperature changes may be quite small. The area amenable to parasitic and vector borne diseases, notably malaria, will expand and impacts could be large. Other direct impacts, such as effects on air pollution, are likely to be smaller. Socially contingent damages to health (via other impacts such as food production, water resources and sea level rise) in vulnerable communities are difficult to estimate but potentially very large.

Agricultural impacts depend upon regional changes in temperature and rainfall, as well as atmospheric carbon dioxide levels. Bio-physical models can identify areas suitable for crops and potential yield changes, but actual yield changes will depend on many factors. Climate variability, as well as mean climate change, is an important consideration. Adaptive responses will be important - choice of crop, development of new cultivars and other technical changes, e.g. irrigation. Impacts of production do not fully determine damages - these will also depend on changes in demand and trade patterns driven by socio-economic factors. The models used in this work take contrasting approaches. In FUND, damages are scaled according to global climate from damage estimates obtained using complex models of world agricultural change. Open Framework damages are based on changes in national agricultural GDP scaled from spatial assessments of land suitable for agriculture.

The literature on water supply impacts includes studies at the regional (catchment level), but there has been only one global scale study. Hydrological simulations can predict changes in water resources. Impacts and damages also depend on demand changes, including those driven by climate change. The water demand of biological systems is affected by various climatic factors, including temperature and humidity. Water supply systems are usually sized to meet (currently) extreme supply/demand conditions and the costs of shortage can be very high. Climatic variability is therefore important in determining damages. The easiest approach to valuation relies on consumer prices to calculate welfare. But in extreme cases, there may also be socially contingent damages.

Sea level rise leads to costs in additional protection, loss of dry land and wetland loss. The balance will depend upon future decisions about what protection is justified. There is no guarantee these will be economically optimal. Costs of protection are relatively well-known, but other costs, in particular valuation of wetland losses, are more uncertain. In addition to these direct costs, land losses will produce migration effects, the costs of which depend on diverse social and political factors.

Impacts on ecosystems and biodiversity are amongst the most complex and difficult to evaluate. Most of the major ecosystem types are likely to be affected, at least in parts of their range. Some isolated systems are particularly at risk. However, there is no comprehensive or reliable assessment of the impacts of climate change on ecosystems. Most valuations rely on *ad hoc* estimates of species loss and contentious valuation studies. The value of ecosystem function may be important, but has received less attention. Even where valuation has be attempted is difficult to apply to marginal changes. There is therefore widespread agreement that accurate valuation of ecosystem impacts of climate change is not possible.

Hazards of extreme weather events raise challenges foTD-0.0Ts8(m2)0.2(f)3ich depend oao811 s30 0 12ah-3 gases (CO_2, CH_4 and N_2O) as a function of discount rate, time of emission and equity weighting assumption. The results of uncertainty analysis are presented for both models. In addition the effects of key sensitivities are examined using FUND.

The marginal damages are calculated for the three major direct greenhouse gases (CO_2, CH_4 and N_2O) as a function of discount rate, time of emission and equity weighting assumption.

Table 6.1 Marginal damages (1995 ECU) of greenhouse gas emissions

Greenhouse Gas	Damage Unit	Marginal Damage from Model			
		FUND		Open Framework	
		1%	3%	1%	3%
Carbon Dioxide, CO_2	ECU/tC	170	70	160	74
	ECU/tCO_2	46	19	44	20
Methane, CH_4	ECU/tCH_4	530	350	400	380
Nitrous Oxide, N_2O	ECU/tN_2O	17 000	6 400	26 000	11 000

Source: FUND v1.6 and Open Framework v2.2
Basis: IPCC IS92a scenario
equity weighted
no socially contingent effects
emissions in 1995-2005
time horizon of damages 2100

For the base case assessment there is close agreement between the results of FUND and the Open Framework (see Table 6.1). Given the differences in model structure and assumptions, this is to some extent fortuitous. Analysis of the breakdown of damages by sector and region shows reasonable agreement in some cases, but divergence in others.

Our analysis indicates that the range of uncertainty is very large. In addition, not all uncertainty is statistical - even the choice of a base case represents a subjective (and often political) view of future economies and societies. Both discount rate and choice of aggregation rule (equity weighting) have large effects on the results. Furthermore, some potentially important issues - socially contingent damages and ecosystem damages are not fully included. The base case values therefore should not be treated or quoted as best estimates.

This assessment has sought to make clear the effects of different assumptions on the marginal damages of climate change. The base case values for carbon dioxide damages calculated from the two models should not therefore be quoted out of context or taken to be a ‘correct’ value. Uncertainty analysis in FUND indicates a geometric standard deviation of approximately 1.8, for uncertainties in climate and impacts that can be parameterised. But many important issues cannot and create additional uncertainty. The effects of some of these sensitivities on the marginal damages of carbon dioxide (calculated in FUND only) are shown in Table 6.2. Assumptions about higher order effects could affect the results even more.

Table 6.2 FUND sensitivity analysis of marginal damages for CO_2 emissions.

Sensitivity	Damages in 1990$/tC (1995 ECU/tC) Discount Rate 1%	Damages in 1990$/tC (1995 ECU/tC) Discount Rate 3%
Base case	170 (170)	60 (66)
No equity weighting	73 (73)	23 (25)
Low Climate sensitivity	100 (100)	35 (39)
High climate sensitivity	320 (320)	110 (120)
IS92d scenario	160 (160)	56 (62)

Source: FUND 1.6
Basis of calculations is our baseline assumptions, i.e.:
damages discounted to 1990;
emissions in 1995-2005:
time horizon: 2100;
no higher order effects.

The valuation of ecosystem and biodiversity impacts of climate change has proved particularly difficult. Ecosystem valuation studies are qualitative or based on *ad hoc* assumptions. Thus, the estimates of values of marginal ecosystem effects that are available are very unreliable. In common with the rest of the ExternE Project no values for ecosystem damages are recommended.

6.6 Conclusions

An approach consistent with sustainability requires consideration of long term impacts, ecosystem stability and scale effects. This suggests the use of an assessment framework in which other approaches than the estimation of marginal damages (as used here) are included. However, damage calculation will remain an important component of any integrated assessment.

The following ranges of estimates were recommended for use within the ExternE National Implementation Study (Table 6.3). It is stressed that the outer range derived is indicative rather than statistical, and is likely to underestimate the true uncertainty. The inner range is composed of the base-case estimates for the 1 and 3% discount rates, and is referred to here as the ‘illustrative restricted range’. There was some debate as to whether the lower bound of this range should be reduced to take account of the 5% discount rate (which would have given a figure of [1995]ECU 8.8/tCO_2) but there was very limited support from the task team for use of the 5% rate. However, the 5% rate was used in derivation of the outer range.

The outer range is based on the results of the sensitivity analysis and the Monte-Carlo analysis of the results of the FUND model. This range varies between the lower end of the 95% confidence interval for a 5% discount rate and the upper end of the 95% confidence for the 1% discount rate. It is referred to as the 'conservative 95% confidence interval', 'conservative' in the sense that the true 95% confidence interval could be broader, because it is not currently possible to consider all sources of uncertainty.

Table 6.3 Recommended global warming damage estimates for use in the ExternE National Implementation Study. The ranges given do not fully account for uncertainty. The derivation of each of the figures identified is described in the text.

	Low	**High**
ECU(1995)/tC		
Conservative 95% confidence interval	14	510
Illustrative restricted range	66	170
ECU(1995)/tCO_2		
Conservative 95% confidence interval	3.8	139
Illustrative restricted range	18	46

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7. ASSESSMENTS OF MAJOR ACCIDENTS

7.1 Introduction

This Chapter presents the conclusions of a larger report (Markandya and Dale [eds.], 1998) produced within the ExternE Core Programme in the period 1996 to 1997. The subject is one of the most difficult to be faced in the project: indeed despite earlier extensive research a clear solution to the problem is still to be identified. The problem is probably most often associated in the western European context with the nuclear fuel cycle, though major accidents are also related to other fuel chains. A key difference for nuclear against fossil technologies is that major accidents for the latter tend to affect mainly those working in mines, on oil rigs, etc. There have been important exceptions to this: the recent pipeline explosion in Nigeria, and the collapse of the spoil heap beside the Aberfan colliery in South Wales in the 1960s which killed 144 people, most of whom were children at school, are obvious examples.

The consequences of serious nuclear accidents are very large and widespread. In monetary terms they could amount to billions of ECU, and there is a high level of public concern (admittedly one which varies from country to country) about the possibility of such an accident. At the same time, expert opinion is that the type of reactors used in Western Europe have a very low probability of the kind of failure that would produce a severe accident. The exact values associated with an event in which there is a failure of containment and hence potentially significant damage vary from one set of experts to another but in general we are looking at probabilities in the order of 10^{-6} and lower. Thus estimated probabilities of this order result in expected damages of thousands of ECU per accident. For many policy-makers, this does not reflect society's willingness to pay to avoid a nuclear accident.[10]

Our earlier report (European Commission, 1995) noted the difficulty of using the expected value of damages in the case of severe nuclear accidents and prepared an addendum to the main report indicating how, in the view of the economists on the team, such issues should be addressed. There were three key questions that needed to be answered.

The first was the need to establish an agreed set of consequences of a nuclear accident and attach probabilities to these consequences. There remains wide divergence in opinion on what consequences should be looked at and hence what probabilities should be attached to those consequences. This implies that we cannot take 'expert' opinion as single-valued and objective, and policy-makers have to choose between different sets of consequences and probabilities.

[10] A further issue concerns the prevalence of wars, which historically have a high frequency, in the order of 10^{-2} to 10^{-1}: in the event of war it is possible that nuclear power plants would be targeted. Against the current unprecedented period of peace and stability in western Europe this appears a remote concern, though it is nonetheless associated with a real, if unquantifiable, possibility.

Related to that is the issue of how one treats public estimates of probabilities versus expert estimates in the assessment of accidents. Clearly both matter; one cannot ignore the careful analysis carried out by the experts, but at the same time one cannot overlook the opinions of the public, whose willingness to pay we are interested in. In the case of accidents which occur with reasonable frequency, this problem is resolved by looking at the relative frequencies of different accidents and basing the probabilities of such accidents on the relative frequencies. For nuclear accidents, there is no such history to draw on. There have been hardly any major incidents with serious consequences in the history of nuclear power; the one at Chernobyl is not considered relevant to the reactors deployed in Western Europe. Hence the divide between public and expert opinion has not narrowed appreciably over time.

The final issue is that account needs to be taken of public aversion to the risk of accidents. The expected value of damages is not enough. The public is willing to pay something for the reduction in risk *per se,* which is not captured in the expected value.

All these points were raised before (European Commission, 1995), reflecting a common opinion of the issues with the US team. A joint paper was published reflecting this opinion (Krupnick, Markandya and Nickell, 1993). Using simple examples of accident scenarios, the paper looked at how the costs of accidents would change once allowance was made for the

The second is the estimation of such costs. Schneider looked at how costs should be estimated and what problems arise in the application of the different methods.

The third is the evaluation of other approaches to estimating nuclear accident costs. Various authors have made attempts to adjust the expected damage approach to allow for risk aversion but, as the paper by Gressmann shows, they are *ad hoc* and without proper empirical and theoretical foundations.

The fourth is the elaboration of the risk-aversion analysis of Krupnick *et al* (1993), taking a more realistic example of a nuclear accident and evaluating a more complex set of associated consequences. This has been done in the paper by Eeckhoudt, Schieber and Schneider, which is important in bringing the risk aversion analysis to a level at which it could be incorporated in an external cost estimate.

The final set of issues is the examination of various alternatives to the expected utility approach and ways in which the issue of expert versus lay probabilities can be analysed more formally and in an integrated manner with the analysis of risk. The paper by Ascari and Bernasconi addresses these issues.

In this summary we go over the main findings from these papers, and draw out the lessons for the valuation of nuclear accidents within an ExternE type framework. To some extent we have made progress so that further quantification is possible. But in some other ways we have opened up the debate, which can only be resolved with further research.

Within the framework of the ExternE project, the cost associated with a nuclear accident was derived on the basis of the economic module available in the COSYMA code (based on the "direct" economic loss associated with health and environmental consequences), including further considerations on the probability of occurrence of the different accidental scenarios as well as specific values for health effects. At that period of time, it was clearly pointed out that this approach was limited and that there was a need for further investigation in order to deal with the risk perception. In this regard, first proposals were presented by Markandya *et al* in the chapter on the valuation of a nuclear accident in the earlier economic valuation report (European Commission, 1995).

The scope of the task presented in this Chapter was to point out some recent results which emerge in the evaluation of the consequences of a severe accident and to make some proposals for taking them into account in the accounting framework methodology developed for the evaluation of the external costs of the fuel cycles.

In order to improve the assessment of severe accidents, the following topics have been considered and are divided into two main parts:

Improvement of the consequences calculations:
- analysis of the relevant accidental scenarios for the evaluation of the consequences and their associated probabilities;
- integration of the indirect costs evaluation;

Integration of the risk perception:
- critical review of practical risk aversion approaches in the external cost studies;
- empirical survey: expert versus expert risk perception;
- expected utility approach;
- economics of risk and uncertainty.

This Chapter contains the contribution of the different teams involved in this task. It does not solve all the issues associated with the calculation of the external costs of a severe accident, but tries to set up the methods available for this calculation as well as to point out the questions concerning the integration of risk perception.

7.2 Expert Probabilities and Lay Probabilities

The paper by Tort reports on the different source terms used in France, Germany and the UK for the analysis of nuclear accidents from a PWR reactor. The key point to emerge from this comparison is the significant differences in the release categories analysed and in the probabilities attached to those releases. The comparison is made most clear by looking at the highest release category in each case. For simplicity only figures for release of caesium are given, as indicative of the more volatile compounds, although there are differences in the relative releases of other compounds in the different source terms. The figures are as follows:

Table 7.1. Release scenarios for different countries (Tort).

Country	Maximum release of Cs (%)	Probability of occurrence	Comment
France	10%	$2\text{-}3.10^{-6}$	Assumed to occur in single release phase without energy. Only 4 release scenarios considered. Next category of accident has only 1% release
Germany	70%	10^{-7}	Six release scenarios included. Next category has 37% release of Cs
UK	50%	$2.4.10^{-9}$	12 release scenarios considered. Next category of accident has 40% release of Cs.

This variation makes cross country comparisons extremely difficult. But, even more importantly, it makes it difficult to accept that there is a unique expert view of the accident probabilities that can be defined as objective.

If the public is presented with such a table it would say, with justification, that the accident scenarios and their associated probabilities are determined partly by judgement and partly by more 'objective' considerations. Hence, it could be argued, a public view of such accident probabilities should also be given some consideration. We return to this point later.

Differences in the professional judgements about nuclear safety are also highlighted in the paper by CIEMAT. Following the approach of Mitchell (1980) and Lindell and Earle (1983), they interviewed different groups of experts in Spain to determine their view on this question. The conclusions are similar to those of the earlier studies. Experts with a greater knowledge of the nuclear sector regard it as a safer source of energy than experts involved in renewable energy development or those dealing with environmental and conventional energy sources. Familiarity diminishes the risk, even for technicians. Unfortunately the conclusions are only qualitative; we do not know how these differences in perception translate into differences in accident probabilities. That would be a useful direction of further research.

7.3 Alternative Approaches to Valuing Risk in Nuclear Accidents

Gressmann reviewed in some detail other approaches to including risk aversion in the assessment of nuclear accidents. The main alternatives are the following:

i. The Ferguson Approach
ii. The Rocard Smets Approach
iii. The Infras/Prognos Study

All of these suffer from not being based on sound empirical and theoretical data. Ferguson assumed that the valuation of the risk increases with the square of the number of people affected in the event of an accident. So an accident with a million people impacted has damages one million times that of an accident with only a thousand people impacted. Furthermore, he states that the cost of an accident increases 'exponentially when global equilibrium is threatened'. There is a claim that these assumptions have been tested empirically but no study has been discovered that reports any such tests. The notion that people have a willingness to pay to avoid accidents with a large number of deaths, or other impacts, that increases more than proportionally with the number of deaths in such an impact is sensible and accords with casual empiricism. But to quantify it in terms of a quadratic function is not valid, unless there is evidence to support such a function. It could equally well be any other power function.

The Rocard Smets approach is to increase the value of an accident by a 'disaster aversion function'. This function multiplies the expected social cost of a nuclear accident by 300 to account for this aversion. The factor of 300 has been derived from some empirical work, but this work is neither cited nor evaluated. Rocard Smets acknowledge that the factor could be as much as 1000. It is stated that these parameters are derived from a comparison of decisions taken by policy-makers. The problem with such an approach is that it assumes rationality on the part of the policy-makers. If we are trying to determine how to make such decisions more rational, it seems tautological to assume rationality in the first place.

Rocard Smets do not assume that the factor of 300 is constant, but that it holds in the case where the number of deaths is about 1000. For more deaths they state that the aversion factor is proportional to the number of deaths to the power of 2/3. Again this is based on public policy and is subject to the same criticisms as the above.

The Infras/Prognos approach is to take the standard deviation of a set of accident probabilities as the measure of the willingness to pay to avoid such an accident. As Gressmann notes, the use of standard deviation to measure the risk is common in portfolio theory, in which the mean and variance (or standard deviation) determine the ranking of a portfolio. The underlying model is that of maximising expected utility, in the case where the utility function is a quadratic one. Unfortunately it is well established that a quadratic utility is not the appropriate one for decision-making under risk. It exhibits sharply decreasing relative risk aversion, which is contrary to the empirical evidence and it is unbounded, which also creates problems when it is applied for changes in income or wealth outside a narrow range.

In fact the actual use of the mean variance analysis by Prognos/Infras is even more bizarre. They ignore the mean and apply only the standard deviation as a measure of willingness to pay. As Gressmann notes, this is not valid, either empirically or theoretically.

These different approaches come up with a very wide range of costs for nuclear accidents. Using the Ferguson approach Pearce estimates externality adders of 0.25-0.625 mECU/kWh. These are about 300 times higher than the ‘average’ expected value estimates of the cost of a nuclear accident. Using the Rocard-Smets approach, Pearce obtains an adder of 3.38 mECU/kWh, which is 2,300 times the expected value estimate. The Infras/Prognos study estimates costs with the risk aversion parameter as between 11.4 and 189.6 mECU/kWh, which is between 22 and 367 times the middle of the range of expected values in their study.

In conclusion, for the future, we would say that such *ad hoc* approaches are to be avoided in analysis of severe nuclear accidents given that some of the assumptions made are highly questionable. This leads to wide variation in range with a limited knowledge of the value of any estimates made.

7.4 Formal Representation of Nuclear Accident Risks Using the Expected Utility Approach

The expected utility approach has the advantage that it is based on proper theoretical foundations and can also draw on a range of empirical work. The earlier Krupnick *et al* paper using this approach is taken further by Eeckhoudt, Schieber and Schneider. They use the constant relative risk aversion function, also used by Krupnick *et al* (1993), namely:

$$U(W) = \{(1-\beta)/\beta\}.W^{\beta}, \text{ with } \beta < 1$$

and

$$U(W) = \ln(W), \text{ with } \beta = 0$$

The first function exhibits positive and decreasing absolute risk aversion while the coefficient of relative risk aversion (A_R) is equivalent to 1-β.

W is household wealth. The risk premium is estimated by calculating the certainty equivalent and a comprehensive numerical application is presented for a nuclear accident affecting a country with a population of 56 million inhabitants. Impacts are divided into two areas: local (less than 100 km from the plant and with 2 million inhabitants), and regional (more than 100 km, with 54 million inhabitants). The consequences of an accident are looked at in terms of food ban costs, individual evacuation and relocation costs, and health and loss of life costs. Probabilities for different consequences are estimated as per the French Scenario ST21, which corresponds to a release of about one percent of the core (this is the next scenario down in seriousness from the one presented in Table 7.1). The coefficient for A_R is taken at 2.0, from a review of various studies of risk aversion. This is also similar to the Krupnick *et al* (1993) study. The authors note that there are some empirical studies that indicate much higher values of A_R, which would imply much greater risk aversion. There are, however, questions as to how valid these studies are as guides to the values of A_R and it seems reasonable to take a value of around 2, although some sensitivity analysis would be useful.

The other key assumption is the initial level of wealth of the individual affected by the accident. The authors take a figure of 2.67 million ECU (MECU). This is made up of a 'value of statistical life' (VOSL) of 2.6 MECU and average financial wealth of 0.07 million ECU. However there are questions as to what is the correct value of a statistical life and there will be significant variations in financial wealth. As Ascari and Bernasconi point out in their paper, it will be incorrect to replace actual values of these variables with the means when the valuation procedure is non-linear. Hence, in any follow-up work one should look at how the overall costs are affected by replacing the averages by a distribution of values for these and lese acatoveralrseralia variese and2po95aia var in amo9ieer2ed se mo.00cct(caia vau in)Tjuoow-up wo e5hei

This function was also looked at by Krupnick *et al* (1993). The coefficients for the parameters A_R and b are similar to those taken by Krupnick *et al* (1993). For A_R they take a mid value of 2.5, against the 2.0 taken by Eeckhoudt *et al*, but also look at a lower value of 0.9. For b the value is selected so that the corresponding coefficient of relative risk aversion (b.W) is equal to the desired value for the mean level of wealth.

The analysis of the risk is also taken from the Krupnick *et al* (1993) study. The adjustment for risk ranges from 1.12 to 2.44 for a risk coefficient of 2.5 and from 1.04 to 1.34 for a coefficient of 0.9. This is similar to the range obtained by Krupnick *et al* (1993) for similar risk coefficients but is lower than the figure obtained by Eeckhoudt *et al* in the above analysis. One reason for the higher adjustment by Eeckhoudt *et al* is that they investigated much smaller probabilities; we know from the earlier work that the smaller the risk of an accident, the greater the divergence between the expected value and the risk adjusted value. A second reason could be that Eeckhoudt *et al* look at more complex risk scenarios, where there is more than one state of the world in the event of an accident. Given the more realistic nature of the Eeckhoudt *et al* scenarios and the lower probabilities they consider, it seems reasonable to conclude that these two pieces of analysis are not inconsistent, but that the Eeckhoudt *et al* figures are a better guide to policy.

Ascari and Bernasconi then go on to look at some further development in risk theory and how they relate to nuclear risks. The first is the model called the Expected Utility with Rank Dependent Probability (EURDP[11]). In this model the probabilities attached to the different outcomes are adjusted according to a probability transformation function. The effect of this adjustment is to give greater weight to events with lower probabilities and lesser weights to events with higher probabilities. This captures the observed phenomenon that people overvalue small probabilities of accidents. It may also capture the fact that lay probabilities of nuclear accidents (which are very small) are cited as higher than expert probabilities. Algebraic forms of the probability transformation function are taken from the risk literature and used to recalculate the ratio of the risk adjusted costs of an accident against the risk unadjusted costs (for the same utility functions as above). The risk adjustment now increases dramatically. For a risk in the range of 10^{-5} it goes up to 141-302 times the unadjusted figure and for a risk in the range of 10^{-6} it goes up to 660-1430 times the unadjusted figure. It is clear that this method gives very large weights to very small probabilities. In the nuclear case it would be interesting to check reported lay probabilities against the adjusted probabilities as given in the adjustment function used by Ascari and Bernasconi.

A second method of modifying the expected utility framework is referred to as the 'disappointment aversion' method (DA). The rationale for this method is that an individual divides his utility from a set of outcomes into two parts: an 'elation' part, where the outcome is better than expected, and a disappointment part, where it is worse than expected. The weights given to the parts depend on the degree of disappointment aversion; the more averse a person is, the greater is the weight, and the more the person will be inclined to select options that avoid such outcomes.

[11] We note that EU theory, though internally consistent, is widely violated in choice experiments (Harless and Camerer, Econometrica, 1994). Further research is clearly necessary here.

As with the EURDP, there is greater willingness to pay to avoid a nuclear risk scenario, where an accident would have a high level of disappointment associated with it. As with EURDP, the disappointment aversion function has been characterised in algebraic form, with quantitative values of the parameters taken from the risk literature. Applied to the nuclear scenarios discussed above, they result in risk adjustments of between 4 and 8.5, much smaller than those obtained with the EURDP. One reason for this is that in the DA method the adjustment varies very little with the probability of the bad event whereas in the EURDP method the probability is the crucial factor.

The debate between these different methods is not resolved in the literature although, as Ascari and Bernasconi note, the opinion seems to be that EURDP is the better method. For the nuclear issue, we would agree with that view. The interpretation of the problem as one of disappointment in the event of an accident seems laboured and the method is not quite as sensitive to either the extent of the loss (as a percentage of income) or the probability of the bad event as one would expect from casual experience and from the application of the conventional expected utility model. Hence, at this stage we would favour the EURDP model.

Given this choice, the question arises as to whether such corrections are justified for the evaluation of nuclear risks? To answer this we have to go back to the key problem identified by Krupnick *et al* (1993) - the ‘disjuncture’ between lay and expert probabilities of nuclear accidents. The lay public do not believe the expert assessments, but at the same time the experts themselves do not have a unique position on these probabilities. The EURDP model seems to pick up the deviation between public and expert probabilities but does not help with the lack of consensus among experts. However, the EURDP model assumes a constant functional relationship between the expert and lay opinions. To some extent these have been changing, although we do not have a careful documentation of how they have changed. In some countries there has been a degree of convergence, in other the disjuncture remains.

What is needed is: (a) a calibration of the probability transformation function so that for the low probabilities the adjustment reflects public perceptions of such accidents, and (b) a procedure for adjusting this function in the light of changes in public perceptions.

As far as (b) is concerned, Ascari and Bernasconi cite the Bayesian procedure proposed by Viscusi, of adjusting public risk evaluations in the light of experience (Prospective Reference Theory). The formula for the final probability is simple enough and one could adjust the probability function sequentially to reflect such changes, but the difficulty is how to incorporate the ‘experience’ component. One way would be to conduct periodic surveys of opinion and use that to revise the prior probabilities. However, no such work has been done to date.

Ascari and Bernasconi then go on to raise two further concerns about the valuation of risks in nuclear accidents. One is the issue of using average values of income and impact for the estimation of risk adjusted damages. With non-linear risk functions there will be errors in using averages. No one knows how large the likely errors are, but this is something that needs to be investigated systematically.

The second is the assumption that one can reduce compound probabilities in which there is a sequence of events to a single probability. For example, if an accident occurs, a person may die immediately. If he or she does not die they may get leukaemia after some time and then die, or they may be ill for some time and recover. The expected utility method calculates a single probability of each such event (immediate death, death after X years, illness, no illness), from the conditional probabilities of each of the events. This assumption (called the reduction of compound lottery axiom) may not be valid. With EURDP, for example, the conditional probabilities each have to have the probability transformation applied. This will also impact on the final estimate of willingness to pay, but we do not have an estimate of how big the impact will be.

7.6 Final Remarks

This Chapter has taken the valuation of nuclear accidents further in a number of respects. We conclude that *ad hoc* rules are not the way forward. Second it has shown that the expected utility framework can be applied to estimating risk-adjusted nuclear accident costs, and that the results are consistent and reasonable. These can offer an immediate correction to the unadjusted costs. Third it has shown that we can incorporate the differences between public and expert assessment of probabilities into a coherent theoretically sound framework. The EURDP model offers a method for doing that, although the precise way in which objective probabilities should be transformed for the nuclear case needs further work.

The Chapter also points to some other areas where further work is needed. First, we need to address the differences in 'objective' probabilities that we observe. In the presence of these it is difficult to know how to encompass the range of views from the experts and link them to the lay probabilities. Second we need to establish a link between measured public probabilities and the expert probabilities, thus identifying the probability transformation function. Third, this function should be dynamic, so that changes in public perceptions are incorporated into a new function, according to the adjustment rule suggested by Viscusi. Fourth, the issue of non-linearity has to be addressed. With risk aversion it is not enough to use average values. We need to look at the distribution of incomes and impacts. Finally, where probabilities are being modified the principle of reducing complex events into simple ones is no longer valid. The time sequence has to be modelled, with the appropriate conditional probabilities.

7.7 References

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8. HEALTH EFFECTS OF PM_{10}, SO_2, NO_X, O_3 AND CO

8.1 Introduction

8.1.1 Purpose

The principal purpose of this chapter is to provide, for the ‘classical’ pollutants, the information on E-R (exposure-response) functions for acute and for chronic effects needed for implementation within the current ExternE programme. This chapter builds on work previously published as part of the EC Joule programme DG XII (ExternE, 1995). When we have updated information, mainly from the large European study; Air Pollution and Health: a European Approach (APHEA), this is indicated in the text.

Following the introduction, Sections 2 and 3 discuss strategic issues by pollutant, and the issues of thresholds and transferability of E-R functions. The next two sections focus principally on which of the E-R functions should be used in the main ExternE quantifications, and which to use in supplementary (sensitivity) analyses. Acute health effects are discussed in Section 4, while chronic impacts are discussed in Section 5. These Sections give little information about the specific E-R functions which have been used; these are described in more detail in Section 6.

8.1.2 Methods

Briefly, we have surveyed the epidemiological literature, in the context of other evidence, to form a view on:

a. What ambient air pollutants have been shown to be *associated* with adverse health effects (acute or chronic), and for what specific endpoints;
b. Which of these associations may reasonably be interpreted as *causal*; (It is important in assessing the effect of *incremental* pollution in ExternE to quantify *causal* relationships, and not just epidemiological associations.)
c. For each relevant combination of pollutant and health outcome where causality, what individual study or meta-analysis gives a representative value for quantification of that effect, taken in isolation;
d. How if at all should the E-R functions from individual studies or meta-analyses be adapted for use in ExternE; and, finally,
e. What combination or set of E-R functions, taken as a unity rather than as a collection of individual functions, provides a basis for quantifying the public health effects of incremental air pollution.

Underlying this methodology is the view that individual E-R functions are components of a wider set of functions; that it is the reliability of the set as a whole which is the principal focus of attention.

Judgements at all of these stages are the focus of debate currently among scientists and policy makers concerned with the health effects of air pollution. In these brief notes we have tried to identify, though not discuss in detail, the most important aspects of that debate. Work in parallel with Ari Rabl at the Ecole des Mines in Paris on how best to describe uncertainty has been carried out and is described in Chapter 5 as well as within this chapter.

An aspect which may appear controversial is d., above: adapting E-R functions for use in ExternE, rather than using directly the E-R functions as published in specific studies. We have taken the viewpoint that our job is not simply to choose a good E-R function from among those published; but, using the published evidence, to provide a good basis for quantifying the adverse health effects of incremental pollution in Europe. In some circumstances (and these are principally to do with transferability) we think that estimates can be improved by adapting available E-R functions rather than by using them directly.

These issues are discussed further below, in the context of specific examples.

8.2 Strategic Issues, by Pollutant

8.2.1 Particles: How should these be represented?

Ambient air pollution particles are a complex mixture, varying in size and in composition. There is very substantial epidemiological evidence of adverse acute health effects of particulate air pollution; and strong, but much less widespread, epidemiological evidence of chronic health effects.

Because particulate air pollution is a complex mixture rather than a single substance, there is a lot of diversity in how particulate air pollution is characterised in various epidemiological

a. The strength of international epidemiological evidence linking various measures of particulate air pollution to acute and chronic health effects;
b. How that evidence relates to particulate air pollution in Europe especially; and
c. What is known about the relative toxicity of various kinds of inhalable particles, and how these relate to the particles being considered in ExternE.

Internationally, there are many studies showing acute health effects of particles expressed as PM_{10} (inhalable particles), or total suspended particles (TSP). In Europe, many studies use black smoke (BS). There are some studies, mostly from North America, showing the effect of finer fractions such as $PM_{2.5}$ or sulphates.

Although causality of acute health effects is now widely accepted, and that of chronic health effects quite widely accepted also, there is no well-established mechanism of action of particulate air pollution. Correspondingly, there is little strong evidence on the relative toxicity of various kinds of inhalable (PM_{10}) particles. There is however some evidence, and strong conjecture, that (per unit mass ambient concentration) the relatively fine fractions ($PM_{2.5}$, sulphates) are associated with greater risks than PM_{10} generally. It may also be that the toxicity of particles is greater according to their acidity, and less according to their solubility.

The particles of interest in ExternE come from four principal sources: primary emissions from power plants; primary emissions from transport; sulphates (from SO_2); nitrates (from NOx). In ExternE 1995, we used E-R functions expressed as PM_{10} in quantifying the effects of all these sources. (Primary emissions from transport were not included.)

In terms of particle size, particles from all four sources lie principally, if not exclusively, within the finer, $PM_{2.5}$, fraction of PM_{10}. Subject to availability of E-R functions, there is therefore a case for using E-R functions for $PM_{2.5}$, or some suitable surrogate, in quantifying the effects of all four sources. This would lead to higher estimated impacts than using functions for PM_{10}.

There is however *a priori* no reason why particles from these diverse sources should have the same toxicity; and there are some relevant compositional differences between sources. In particular, nitrate particles are relatively highly soluble, and the mullite particles emitted from power plants may in compositional terms be relatively inert. Also, there is no compelling reason to use the same E-R functions for all four sources. Finally, E-R functions for $PM_{2.5}$ are generally less well established than for PM_{10}. Indeed, there are no suitable European E-R functions directly in terms of $PM_{2.5}$; and so we need either to use a surrogate sub-fraction such as black smoke, or transfer E-R functions from North American studies. Either approach involves important approximations or uncertainties.

Against this background, we propose a compromise. We recommend that for the main implementation we use (per unit mass of incremental particulate air pollution) E-R functions for particles indexed differently according to the source, as follows:

Primary source, Power station:	PM_{10}
Primary source, Transport:	BS/$PM_{2.5}$
Sulphates:	BS/$PM_{2.5}$
Nitrates:	PM_{10}

In addition, the E-R functions can be applied as follows for sensitivity analysis:
i. expressed as PM_{10};
ii. expressed as BS/$PM_{2.5}$;
for particulate pollution from all sources.

8.2.2 Ozone

Quantifiable acute health effects, additive to those of particles

ExternE 1995 included a set of E-R functions linking ambient ozone with a wide range of acute health effect endpoints. The epidemiological evidence at that time (see Methodology Report, Ch 4) was strong on the effects of ozone in some locations, especially in North-Central America (California, USA; North-East USA/ South East Canada; Mexico City).

A major unresolved issue was whether ozone effects were identifiable, or even occurred, in other situations; though a series (1995) of hospital admission studies from various US locations suggested that they did.

The APHEA results have confirmed a relationship of ambient ozone with acute mortality and hospital admissions in Europe. The overall evidence strongly supports the view that the acute health effects of ozone can and should be quantified; and that the estimated health impacts from ozone should be added to those of particles.

How should daily ozone be represented?

Although ozone is not a mixture in the same way that particles are, epidemiological studies represent the effects of daily ozone characterised in various ways; e.g. as 1-hr daily max.; as 5-hr daily average; as 8-hr. daily average; as 24-hr daily average.

For ExternE 1995, we used E-R functions expressed in terms of 1-hr daily max. This is principally because the early strong evidence of ozone effects came from human experimental (chamber) studies, with short-term exposures; and the early epidemiology was mostly in terms of daily 1-hr max. The more recent epidemiological studies have tended to consider longer time-frames, on the growing understanding that these may be more relevant biologically.

Now, in order to link with the modelling of incremental pollution, we give E-R functions expressed in terms of 6-hr daily average. This involves applying conversion factors to functions derived originally using different indices. Data from some of the European APHEA-study cities suggest a conversion factor of 6-hr daily average ozone as approximately 0.8 times 1-hr daily max; and so an E-R function, expressed as percent change in health effect per ppb of 1-hr max ozone, should be scaled upwards by a factor of 1.25 to give the equivalent percent change per ppb of 6-hr daily average.

8.2.3 SO_2: There is an association, but is it causal?

There is an association

In ExternE 1995, we did not quantify any health effects associated with SO_2 as a gas. We recognised (Methodology Report, Chapter 4) that this was a difficult decision: the evidence at that time was ambiguous. We did however quantify effects associated with sulphates.

Further evidence on the acute health effects associated with SO_2 is now available, principally from the European APHEA studies, also from the HEI-sponsored re-analyses of data from some US cities. The US re-analyses strengthened the case for an association between daily ambient SO_2 and acute health effects. With APHEA, it is fair to say that an association with acute mortality is well-established; and probably one with hospital admissions also. So it is possible, using APHEA, to provide E-R functions linking ambient SO_2 with the most severe acute health endpoints.

But is it causal?

There is however still a major difficulty in interpreting whether these functions represent an association which is causal. The principal alternative viewpoint is that SO_2 in these studies is acting as a surrogate for other pollutants, especially fine particles (e.g. sulphates) not well quantified in the particle measurements available for study.

It is important for us in ExternE to quantify causal relationships, and not just epidemiological associations. So there are still some major unresolved issues about whether or not to quantify an SO_2 effect.

In the APHEA studies, the size of the apparent SO_2 effect does not depend on the background concentrations of ambient particles. In the context of the evidence as a whole, including this result, we recommend that the functions for SO_2 are used in the main ExternE implementations now; and that the estimated impacts are added to the effects of particles and of ozone.

We suggest sensitivity analyses in which SO_2 is excluded.

8.2.4 CO

There is relatively little epidemiological evidence concerning CO, so that it is difficult to place in context the results from a few (well-conducted) studies which report positive associations. Those studies do provide the basis for E-R functions, but they do not give strong guidance on how representative or transferable these functions are. Specifically, whereas in many studies CO is not examined as a possibly causative pollutant, there are also well-conducted studies which do consider CO and yet do not find a CO-related effect.

On present evidence, we recommend that the functions for CO and acute hospital admissions for congestive heart failure are used; but that functions for CO and acute hospital admissions for ischaemic heart disease and acute mortality are only used for sensitivity analyses. In epidemiological studies the effect of CO are represented as 1-hr daily maximum and 8-hr daily average. Consequently, E-R function expressed in terms of 1-hr max CO and 8-hr daily average have been scaled up by a factor of 2 and 1.5 respectively.

8.2.5 NO_2

In ExternE 1995, the epidemiological evidence regarding NO_2 was assessed. Some studies reported NO_2 effects. However, the broad thrust of the evidence then was that apparent NO_2 effects were best understood not as causal, but as NO_2 being a surrogate for some mixture of (traffic-related) pollution. We concluded that a direct effect of NO_2 should not be quantified. (Indirectly, NOx did contribute, as a precursor to nitrates and to ozone).

The APHEA results report positive associations between NO_2 and daily mortality or respiratory hospital admissions in several European cities. Consequently, it is possible to propose E-R functions. APHEA however also supports the view that the apparent NO_2 effects may be due to particles; or at least, are highly dependent on background particle levels. Against this background, we recommend that the E-R relationships for NO_2 are not used, except for sensitivity analyses. In the epidemiological studies from APHEA the impacts of NO_2 are described in terms of 1-hr maximum. Consequently, the E-R functions in terms of 1-hr maximum have been scaled up by a factor of 1.67 to express results in the metric of 24-hr average.

8.3 Other Overall Strategic Issues: Transferability and Thresholds

8.3.1 Different functions in `Western' and `Eastern' Europe

For particles (black smoke) and for SO_2, the APHEA results for acute mortality cover a sufficient diversity of European cities that it has been possible to check if the same E-R functions are applicable throughout Europe. APHEA results show important differences between estimated effects in `Western' and `Eastern' Europe, with percent change per unit exposure lower in the East than in the West.

We are not sure why these differences within Europe occur; and in particular how specific they may be to acute mortality rather than to other endpoints. There are not corresponding data for other endpoints. What counts here as `East' and `West' Europe is not clear cut. The APHEA evidence on East, or East-Central, Europe is based on four cities in Poland (Cracow, Poznan, Wroclaw, and Lodz), and on one (Bratislava) in Slovakia; and the differential effect was found principally in the Polish cities. The `Western' cities in the acute mortality comparisons of APHEA are Barcelona, Lyon, Paris, Athens, Köln, Milan, and London; i.e. these include cities in Southern Europe also. Acute mortality results are not reported within APHEA from Helsinki, the only Scandinavian city in the study; but its results on acute hospital admissions are consistent with that of other `Western' cities.

Non-APHEA studies of acute mortality from East-Central Europe include that by Spix *et al* (1993) from Erfurt in (East) Germany; and the Swiss three-city study of Wietlisbach *et al* (1996). Results from Erfurt fit better with the `Eastern' rather than the `Western' results from APHEA; whereas the Swiss results fit better with the APHEA Western findings.

Finally, the current version of the ECOSENSE implementation programme was not developed with a view to implementing different E-R functions for different parts of Europe; though such a development is possible in the future.

To estimate the influence of applying separate E-R functions for Eastern and Western Europe, these were implemented for one of the German case studies. In view of the distance and the prevailing wind direction, pollutant emissions in Germany should be the most relevant for Eastern Europe. The application of the Western functions to all Europe resulted in an overestimation of the impacts of 1.4%. Because of these difficulties, and because most of the population at risk and incremental pollution is in the ‘West’ European area, we have used ‘West’ European functions during the present phase of ExternE.

There is no evidence on whether the effects of ozone and, if real, of CO and of NO_2, vary by East-West Europe. The E-R functions proposed come from Western European cities.

8.3.2 Choice of European functions

At the time of working on the ExternE project the meta-analyses from the APHEA study were not available. We relied on published reports of the individual APHEA cities, to form a judgement on the E-R functions which would be representative of a European setting. Occasionally we used published reports of a study which was not part of the APHEA study, but which was set in an APHEA or other European city. For example, the study of Verhoeff *et al* (1996) was set in Amsterdam (an APHEA city), but did not come under the umbrella of the APHEA study.

8.3.3 Transferring exposure-response functions from North America

The quantification for ExternE 1995 was highly dependent on results from epidemiological studies carried out in North America. We are now in the substantially more favourable position that, for severe acute health effects (mortality and hospital usage), there are European data, principally from APHEA. The international evidence, of which that from North America still plays a central part, remains very important in influencing judgements on the reliability of associations, and on causality. But where practicable, however, we have based exposure-response functions on European studies.

Another advantage of having substantially more European evidence is that, for some pollutants and for some endpoints, we can now assess how similar are the results in North America and in Europe; and, by implication, assess the transferability to Europe of North American results.

There are differences. For acute mortality, where the evidence is strongest, the estimated effect of particles is generally lower in Europe than in North America. The reasons are not well established; but they may be because of higher co-exposures to SO_2 in Europe. (There is some evidence in US results of higher particle effects where SO_2 is low). For particles (PM_{10}), the new estimates for hospital admissions in Western Europe are about one-half of those estimated for ExternE, using US data. (There were much higher percentage increases in hospital admissions in the US, but with lower background rates than in Europe).

The situation regarding ozone is less consistent. The estimated effect of ozone on acute mortality is much higher in Europe than the US. However, there is not a wide base of evidence from North America on acute mortality and ozone; what is available is from high-ozone regions. Regarding respiratory hospital admissions, the new APHEA-based estimates for ozone (6-hr mean) are slightly higher than the North American results used in ExternE 1995, but the relative difference is not nearly so great as for mortality.

Identifying these differences is important because for other endpoints there are no suitable European studies; and so any quantification must be based on studies conducted elsewhere, and typically in the USA or Canada. This is crucially the situation for the chronic health effects of particles, whose impact on the overall evaluation is very large.

In these circumstances, we have sometimes scaled the US or Canadian exposure-response functions (scaled downwards for particles, upwards for ozone) for use in Europe. The magnitude of appropriate scaling is to some extent arbitrary and certainly is arguable, because it is impossible to know what is the `right' scaling factor. What we can assess, however, is whether it is possible to improve on no scaling at all, i.e. on an implied scaling factor of one; and that has been the basis for our judgements.

8.3.4 Using local functions

It is possible, because of the many studies which have now been carried out in Europe, for the case studies to derive exposure-response functions for at least some endpoints from studies conducted locally. The question of which is more appropriate then arises, the general exposure-response functions presented here, or local functions.

It is intended that for the main implementation, every team uses the same standard set of exposure-response functions. This will allow comparability (between locations, between fuel cycles) using a common system. We would encourage diversity in supplementary analyses. In particular, now that epidemiological results are available from many European cities, it might be an advantage if some teams can carry out sensitivity analyses using local results.

8.3.5 Conversion from ppb to gravimetric units

In the tables in order to express all functions per unit pollutant, in gravimetric units, the exposure-response functions are expressed in $\mu g/m^3$. The conversion factors used were:

1 ppb O_3 = 1.997 $\mu g/m^3$ of O_3;

1 ppb NO_2 = 1.913 $\mu g/m^3$ of NO_2;

1 ppm CO = 1.165 mg/m^3 of CO.

8.3.6 Exposure-response functions for $PM_{2.5}$

At present, epidemiological studies examining the relationships between particles expressed as $PM_{2.5}$ and health come almost entirely from studies in North America. This does not as yet constitute a comprehensive body of evidence for quantification. Also, as noted above, there is a problem of scaling in transferring exposure-response functions for acute effects from North America to Europe. Consequently, we have not used the North American functions for $PM_{2.5}$.

We considered using BS as a surrogate for $PM_{2.5}$ in Europe in order to give access to the APHEA results for quantifying $PM_{2.5}$ effects. Both BS and $PM_{2.5}$ are similar as a fraction of PM_{10}. They do however represent different sub-fractions and the BS-$PM_{2.5}$ relationship may vary considerably by site, climate and other circumstance. We decided therefore that this indirect approach to estimating a $PM_{2.5}$ effect was too unsound.

We chose a third option, to convert from PM_{10}. Results quoted in tables for $PM_{2.5}$ have been directly converted from the exposure-response functions for PM_{10} by multiplying by 1.67. This conversion factor is based on Dockery and Pope (1994) where it was derived principally from the US experience.

There is some indirect European evidence that this is a reasonable factor to use in Europe also, assuming BS does correspond to $PM_{2.5}$ for acute mortality, the exposure-response function for BS from Sunyer *et al* (1996) is approximately 1.67 times the exposure-response function for PM_{10} from the APHEA study.

8.3.7 Thresholds

For many of these pollutants, there clearly is a threshold at the individual level, in the sense that most people are not realistically at risk of severe acute health effects at current background levels of air pollution. There is however no good evidence of a threshold at the population level; i.e. it appears that, for a large population even at low background concentrations, some vulnerable people are exposed some of the time to concentrations which do have an adverse effect.

This understanding first grew in the context of ambient particles, where the `no threshold' concept is now quite well established as a basis for understanding and for policy.

For ExternE (1995), our understanding of the epidemiological evidence on ozone was that it did not point to a threshold. The situation was unclear however, and the limited quantification of ozone effects did include a threshold. This, however, was principally because of difficulties in ozone modelling, rather than on the basis of epidemiology as such.
Overall, the APHEA results do not point to a threshold for the acute effects of ozone; and we understand that the World Health Organisation (WHO) is now adopting the `no threshold' position for ozone as well as for particles. Against this background, we recommend that quantification of all health effects for ExternE now be on a `no-threshold' basis.

8.3.8 Additivity

Within endpoint, effects are additive unless otherwise stated.

Between endpoints for particles, assume that all respiratory hospital admissions (RHA), congestive heart failure admissions (CHF), and cerebrovascular admissions (CVA) also involve restricted activity days (RAD).

In adjusting RADs to take account of hospital admissions, it is arguable whether or not to convert each admission into equivalent hospital days. On balance, we have decided to do so using approximate average length of stay.

This gives:

$$\text{Net RAD} = \text{RAD} - (\text{RHA} * 10) - (\text{CHF} * 7) - (\text{CVA} * 45).$$

The final results are insensitive to which adjustment has been made. Indeed, with the given assumptions on the length of stay, the Net RAD are 98.6% of the RAD, i.e. the adjustment is negligible in view of the uncertainties involved in the assessment.

Between endpoints for ozone, all asthma attack days (AA) are also MRADs. The function for MRADs refers to adults (80%) only, while that of AA refers to all asthmatics (3.5%). Hence

$$\text{Net MRAD} = \text{MRAD} - (\text{AA} * 0.8 * 0.035).$$

8.3.9 Representing uncertainty

After many discussions, led by Ari Rabl, a consensus has emerged to use uncertainty labels for each impact, somewhat analogous to the H, M and L confidence levels used in the first series of ExternE reports (EC 1995). However, we now use a more quantitative definition based on geometric standard deviations σ_G and confidence intervals of the lognormal distribution. The main reasons for this decision are:

i) we can make meaningful distinctions between different levels of confidence, in other words we are confident about the order of magnitude of the uncertainties;
ii) the identification of uncertainties and the specification of the probability distributions of some of the elements of the damage cost calculations remain elusive;
iii) providing more explicit values of σ_G for each impact under such circumstances would imply a false sense of precision ('precisely wrong rather than approximately right');
iv) the labels that we have chosen are intuitive and convey the key message even to readers without training in probability theory, yet they do allow a quantitative interpretation.

The labels are:

A = high confidence, roughly corresponding to σ_G = 2.5 to 4;
B = medium confidence, roughly corresponding to σ_G = 4 to 6;
C = low confidence, roughly corresponding to σ_G = 6 to 12.

In some cases a '?' is added to indicate cases where the damage cost may well be lower than our estimate. For example, while mortality from PM_{10} is assigned a label B, mortality from nitrates is assigned a 'B?' because it has been obtained by applying the PM_{10} exposure-response function as if nitrates had the same toxicity as particles (reasoning by analogy without explicit epidemiological evidence). The ? is also added to impacts such as mortality from CO and asthma attacks from O_3 were the evidence for the E-R function is weak, i.e., the end points that have been classified 'for sensitivity analysis' in the tables from IOM (Hurley *et al*, 1997).

The label '!', used in the first EC (1995) for 'extremely uncertain' has been dropped. We believe that the label C is sufficiently ample, allowing a 95% confidence interval that spans more than four orders of magnitude.

8.3.10 Implementation/ use of the E-R functions given below

In terms of implementation, the various functions can be used more-or-less directly, by appropriate linkage with incremental pollution and relevant population-at-risk. The principal exception concerns chronic mortality, where more complex intermediate calculations, based on life-table methods, are needed.

Briefly, the basic inputs are the E-R function from Pope *et al* (1995), expressed as a percentage increase; the age-specific death rates (hazards) from the baseline population in whom effects are being estimated; the age structure of the baseline population; and computer programs to carry out life table calculations and to summarise the results suitably. Life tables are constructed under two scenarios: the first where there is no change to baseline hazard rates and the second where the hazard is perturbed by the percentage increase from Pope *et al* (1995) relevant to unit incremental pollution ($\mu g/m^3$ PM_{10} or $PM_{2.5}$). Under each scenario, the total years of life survived can then be summarised, e.g. overall, over various time-periods, for those in various age-groups currently. Differences in survival time are the years of life lost (YOLL) attributable to the pollution increment.

Aggregate YOLL was remarkably stable in implementations using various European populations-at-risk and time-lags between exposure increment and effect on age-specific death rates. For example, the total YOLL can be expressed as an average life shortening per individual of 0.001 years, for a 1 $\mu g/m^3$ increment of $PM_{2.5}$ sustained for one year only. Thus, the cohort studies lead to information about aggregate Years of Life Lost (YOLL) in a population, and not about `extra deaths'. (In the long run, there are no extra deaths).

8.4 Evaluation of Acute Effects by Endpoint and Pollutant

The next two sections discuss the reasons underlying the choice of E-R functions and we give an indication of which functions should be used in the main implementation and which should be considered for sensitivity analyses only.

8.4.1 Acute Effects on Mortality

We have more information about acute mortality than about any other endpoint.

Particles

Table 8.1 gives the E-R function for particles in terms of PM_{10} and $PM_{2.5}$.

In choosing an E-R function in terms of PM_{10}, note that in the APHEA study there were no direct analyses of acute mortality in relation to PM_{10}. However, some cities examined indices of particles which are close to PM_{10} and may be taken as equivalent: mortality analyses from Paris and Lyon examined PM_{13}, while in Köln, PM_7 was measured. Also, separate (non-APHEA) reports on acute mortality and air pollution in Amsterdam (Verhoeff *et al,* 1996) and in Birmingham (Wordley *et al,* in press) give E-R relationships directly in terms of PM_{10}. In addition, TSP was measured in Barcelona, Bratislava, Milan and (on some days only) in Köln. E-R functions in terms of PM_{10} can be derived using the US conversion formula, PM_{10} = TSP * 0.55 (Dockery and Pope, 1994). The Köln study (Spix *et al,* 1996) reports mean values for PM_7 and for TSP of 34µg/m^3 and 68 µg/m^3 respectively, suggesting that this conversion factor may be applicable in Europe also. (Note however that in the Köln study, the difference in regression coefficients between TSP and PM_7 was small).

In the light of these results, we have derived a function for PM_{10} using the mean of the effects of PM_{10} as estimated from the Köln and Amsterdam data. The recent publication (Katsouyanni *et al,* 1997) of the meta-analysis of acute mortality and particulates gave an identical result.

For $PM_{2.5}$, we have scaled the same E-R function by a factor of 1.67.

These European results for particles are lower than the ExternE 1995 results transferred from the USA. We are not sure why; it may be because of higher associated SO_2: see above. In practice, for acute mortality the new lower estimates are partly offset by inclusion now of SO_2 estimates; though the effect of this will vary by application, e.g. by fuel cycle.

Ozone and SO_2 (Include, with particles, in main analyses)

Table 8.1 gives E-R functions for ozone and for SO_2. These are to be included in the main analyses. Take it that the impacts for particles (from various sources), from ozone and from SO_2 are additive. Note however that the causality of SO_2 is uncertain. It would be useful to comment on what difference it would make if this (and later) SO_2 functions were omitted.

Here and elsewhere, in selecting individual E-R functions when results from several cities or regions are available, we have not carried out a meta-analysis proper. (The APHEA authors have a series of meta-analysis papers in preparation). Instead, we have chosen a `typical' or `representative' E-R function; i.e. one whose value is close to the average of those available. Often this is the E-R function from one city. In a few instances, it is the unweighted average of estimates from two cities, usually one with `high` effect, the other with `low' effect. The estimates from these approaches would be similar to that from a proper meta-analysis.

From the viewpoint of implementation, for ozone, Table 8.1 gives the E-R function from Sunyer *et al* (1996) scaled upwards for 6-hr daily mean. For SO_2, the E-R function is the mean from London and Athens.

Note that for ozone, results are available from studies in `Western' European cities only; and that the estimated effects of ozone are much higher than those implied by the US E-R function used in ExternE 1995.

CO and NO_2 (Include in sensitivity analyses only)

Table 8.2 gives acute mortality E-R functions for CO and for NO_2. These are not for the main implementation: they are for sensitivity analyses only.

The metric for CO used by Touloumi *et al* (1994) was 8-hr maximum and so was scaled upwards by a factor of 1.5 to give 24-hr mean.

The NO_2 function is the mean of results from Barcelona (which gave relatively high RRs for NO_2 and mortality) and London (relatively low RRs). The relationship was expressed in terms of 1-hr max., because the Barcelona results are given in this metric only and then scaled upwards, for linkage with 24-hr incremental NO_2. The ratio of 1-hr max to 24-hr average NO_2 was 1.54 from London data, 1.64 from Paris data, 1.89 from Lyon, and 1.6 from Köln,

Table 8.1 gives E-R functions for particles in terms of PM_{10}. Relevant results were available from APHEA for Western European cities only. The E-R functions of Table 8.1 are from the Paris data (Dab *et al*, 1996) where PM_{13} was analysed, and we have taken PM_{13} as equivalent to PM_{10}. In Paris, for respiratory hospital admissions, the estimated effects per µg/m^3 BS were very similar to those per µg/m^3 PM_{13}. (We would have expected the BS effect to be the higher, as for acute mortality). In the Paris study, the mean value of 24-hr BS was 40 µg/m^3 compared with an average 24-hr PM_{13} of almost 55 µg/m^3, giving a ratio BS to PM_{13} of 0.73.

Table 8.1 gives E-R functions separately for ozone (1-hr daily max. scaled upwards to give 6-hr daily mean), for SO_2 and for NO_2 (1-hr daily max. scaled upwards to give 24-hr daily mean). Similar comments apply as for acute mortality. In particular, estimates for fine particles ($PM_{2.5}$) have been derived by scaling upwards by a factor of 1.67, the estimates for PM_{10}. Impacts for particles, for ozone and for SO_2 are treated as additive. NO_2 was used only for sensitivity analyses and not for the main implementation.

There are no E-R functions linking CO and respiratory hospital admissions.

Cardiovascular and cerebrovascular hospital admissions

The acute mortality deaths associated with air pollution are from (non-malignant) cardio-respiratory causes. It is logical therefore to expect pollution-related increases in cardiovascular as well as in respiratory hospital admissions.

This endpoint was not included in ExternE 1995 because there were no suitable E-R functions at that time.

Morris *et al* (1995) studied daily hospital admissions for congestive heart failure in relation to a range of pollutants, notably NO_2, SO_2, CO and ozone, but not particles, in people aged 65+ in seven US cities. In multi-pollutant models they reported an association with CO which was statistically significant at five of the seven cities studied; but no corresponding association with any of the other gases studied. This is an important finding, partly because a relationship with CO is plausible biologically; but it is of limited benefit for quantification because particles were not included also.

Schwartz and Morris (1995) reported results in greater detail from one of these cities, Detroit. This study again focused on daily hospital admissions among those aged 65 or more; but examined PM_{10} as well as ozone, SO_2 and CO and considered ischaemic heart disease (IHD) as well as congestive heart failure. Results from two-pollutant models, taking due account of longer-term trends and cycles and of climate, showed statistically significant associations between congestive heart failure and both PM_{10} and CO. Adjustment for either of these pollutants had little effect on the estimated impact of the other; i.e. the estimated effects of particles and CO were relatively independent; and by implication were additive also. Both SO_2 and CO, as well as PM_{10}, individually showed statistically significant associations with IHD; but only PM_{10} remained clearly related to IHD in two-pollutant models.

Burnett *et al* (1995), in a follow-up to Burnett *et al* (1994), studied cardiac as well as respiratory hospital admissions over six years (1983-88) at 168 acute care hospitals in Ontario, Canada. Pollutants considered were particles (sulphates) and ozone. Overall, ozone was not related to cardiac hospital admissions. Particles (sulphates) were related, however. Burnett *et al* had studied people of all ages; separate analyses of those under and over 65 years of age showed relatively small differences in estimated effects (3.5% increase per 13 $\mu g/m^3$ sulphates in those aged 65+; 2.5% in those aged under 65). After suitable scaling, including conversion to PM_{10} as in Dockery and Pope (1994) (sulphates = 0.25 PM_{10}), these estimates imply a similar percent increase per $\mu g/m^3$ as Schwartz and Morris (1995); but with the major difference that they apply to younger people also.

Wordley *et al* (in press) examined daily levels of PM_{10} in relation to a wide range of endpoints, including daily hospital admissions for IHD and for cerebrovascular disease, among people in Birmingham, England. Having adjusted for temporal patterns and for climate, there was no good evidence of a relationship between IHD admissions and daily PM_{10}. There was however a statistically significant relationship between acute cerebrovascular admissions and same-day (but not earlier) PM_{10}.

From the viewpoint of quantification for ExternE, Table 8.1 includes E-R functions from Schwartz and Morris (1995) linking particles (PM_{10}) and CO with two different types of cardiovascular condition: congestive heart failure and ischaemic heart disease. Table 8.1 links cerebrovascular hospital admissions and particles, again as PM_{10}, from Wordley *et al* (in press).

The E-R functions from Schwartz and Morris (1995) refer to people aged 65 years or more only. Following Burnett *et al* (1995), discussed above, it seems that the effects occur in younger people also. However, in the Ontario study 62% of all acute cardiovascular admissions occurred among people aged 65 or more; so that the shortfall in estimated effect by ignoring the under-65s is not great. The function from Wordley *et al* refers to all ages.

The functions for congestive heart failure (Table 8.1) and for cerebrovascular hospital admissions (Table 8.1) should be used in the main analyses; but those for IHD (Table 8.2) should be included in sensitivity analyses only, because of the negative results on IHD and particles from Wordley *et al*, and the lack of statistical significance of the CO effect on IHD in Schwartz and Morris (1995), after adjustment for particles.

In implementing these E-R functions note that for consistency, we would scale down the estimated particle effects from Schwartz and Morris (1995), for application to Europe. However, we have not done this, with the effect of informally balancing a probable over-estimation by applying US functions against the under-estimation by ignoring under-65s. For fine particles, there are no studies giving results in terms of $PM_{2.5}$ and so for $PM_{2.5}$ estimates the PM_{10} E-R estimates were scaled upwards by a factor of 1.67.

The CO relationships from Schwartz and Morris (1995) are in terms of daily 1-hr max CO. There is no information from APHEA to give a 1-hr max to 24-hr conversion. Direct application of the given CO functions to incremental pollution expressed as 24-hr daily average will under-estimate the effects; and so compound an under-estimation by ignoring those aged under 65. We have scaled the CO functions in Tables 1 and 2 upwards by a factor of two to compensate for both types of under-estimation.

There is inadequate information for any re-scaling of CO effects, either in transferring from US to Europe, or within Europe.

8.4.3 Other acute health endpoints

What endpoints?

From here on, that is, for less severe acute effects than mortality and hospital usage, there are data for respiratory acute health endpoints only and not for cardiovascular endpoints.

What pollutants?

For these other acute health endpoints, we propose functions for particles and for ozone only (as in ExternE 1995); and not for SO_2, for NO_2 or for CO. Though logically, we might expect some effects, e.g. of SO_2 on asthmatics.

8.4.4 Emergency Room Visits (ERVs)

Choice of endpoints and of source papers

As with hospitals admissions, for emergency room visits (ERVs) we looked for, and used, E-R 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 E-R 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:

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 $\mu g/m^3$ PM_{10} per 100,000 people at risk.

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).

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.

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).

This is a difficult endpoint, in that it reflects North American rather than European hospital usage. We have recommended the same E-R functions, for PM_{10} and for ozone, as for ExternE 1995. These are given in Table 8.2 and are to be used only for sensitivity analyses.

The E-R functions for particles are from European studies, so the issue of transferability from the US does not arise.

For fine particles ($PM_{2.5}$) the PM_{10} values were scaled upwards by a factor of 1.67.

The ozone results are based on a US study. There is perhaps a case for scaling these estimates upwards, given higher ozone effects in Europe. However, adjustment for particles (PM_{10}) in the original paper by Cody *et al* (1992) was poor (see Methodology Report, p123). Because of this, and because of higher ERV usage in North America, the values may be high in any case; so we have not re-scaled upwards.

The daily ozone data in Cody *et al* (1992) refer to a specified 5-hr period each day. We have used unchanged the E-R functions from the original paper as estimates of effects based on 6-hr daily average ozone, as needed for ExternE.

8.4.5 Restricted Activity Days (RADs)

Definition

One measure of morbidity, important in economic valuation though relatively rarely used in epidemiology, is a restricted activity day (RAD). An RAD is defined as a day when a study subject was forced to alter his or her normal activity. An example of a severe restriction is a day when it was necessary to stay in bed because of ill-health. RADs include days off work for employed adults and days off school for children, whether or not the subject was confined to bed on those days. Minor Restricted Activity Days (MRADs) do not involve work loss or bed disability, but do include some limitation on 'normal' activity.

The US Health Interview Study (HIS)

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 and is based 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 it is necessary to adjust in the analyses for possible confounding factors such as age, race, sex, income and education, information on these factors being available for each individual subject. Thus, though the HIS studies investigate acute effects, in terms of design they are very different from the time series and panel studies on which most of the E-R functions for acute effects have been based. In design terms they resemble rather the studies of longer-term effects described later.

Information on smoking habits is not available generally and this is an important limitation.

Choice of papers

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). All these papers focus on the number of relevant RADs in the two-week period prior to interview.

Portney and Mullahy considered data from 1979 only, because supplemental data were available on the smoking habits and residence of a subsample of the 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) considered only fine particles (FP, i.e. $PM_{2.5}$) estimated from airport visibility data, but examined 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.

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, its population is more representative, even though that representativeness may have complicated the determination of number of RADs.

E-R functions

We know of no major new information about RADs in recent years and so we have not attempted to improve on the ExternE (1995) E-R functions. Because in the original papers on RADs (Ostro, 1987; Ostro and Rothschild, 1989) particles were studied as 'fine particles', i.e. as $PM_{2.5}$, the values for % change per $\mu g.m^{-3}$ PM were scaled down (by a factor of 0.6) to give estimates as PM_{10}. The estimates for particles have been scaled *downwards* by a further factor of two (i.e. to one-half) for European implementation. We have not re-scaled the ozone estimates.

The E-R functions are give in Table 8.1. These include E-R functions in terms of $PM_{2.5}$, derived directly from the studies, but also scaled downwards to one-half their original values, for a European implementation.

8.4.6 Acute effects in asthmatics: particles

In ExternE 1995 we estimated the increase in `asthma attacks' in asthmatics, related to incremental increase in particles ($PM_{10}$) and ozone. `Asthma attack' is not a well-defined endpoint, in the sense that criteria for what constitutes an asthma attack are not consistent across studies. For particles, `asthma attack' as used in ExternE 1995 was in effect `shortness of breath' in asthmatics.

For particles, we now propose some different endpoints for asthmatics:
a. Increased use of medication (bronchodilator usage);
b. Increase in respiratory symptoms.

Bronchodilator usage

The E-R functions (Table 8.1) are from two European studies of asthmatics, separately for children and for adults. Note that these studies are based on small numbers of subjects, in only one European country (Netherlands); and so there may be important problems of representativeness.

Estimated effects per asthmatic person per day are higher in adults than in children. In fact there was a lower percentage increase in adults, but baseline rates were much higher in adult asthmatics (43%) compared to children (10%).

The functions for $PM_{2.5}$ (Table 8.1) were obtained by scaling the PM_{10} values upwards by a factor of 1.67.

Lower respiratory symptoms, principally wheeze

Table 8.1 gives functions for PM_{10} and for $PM_{2.5}$, separately for young people and for adults, from European studies; i.e. the question of scaling for US studies does not arise.

Cough

Table 8.1 gives, again separately for children and for adults, E-R functions for increased cough days in asthmatics. As before, the functions for $PM_{2.5}$ were obtained by scaling the PM_{10} values upwards.

The E-R functions for adults come from a European study. Those for children are from the US, and we have scaled downwards the functions for children by a factor of two.
(An implication will be that there will then be no important difference in estimated effect between adults and children).

8.4.7 Acute Effects in Asthmatics: ozone

There is some uncertainty about whether the evidence overall supports quantification of this endpoint. The E-R function from Holguin *et al* (1984) is based on a small study, and gives high estimates of effects (ExternE, 1995). The function from Whittemore and Korn (1980) gave more conservative results. We recommend that the Whittemore and Korn (1980) function be used, adapted for 6-hr daily ozone.

8.4.8 Respiratory Symptoms in the General Population and Particles

In considering respiratory symptoms in the general population, we can either find suitable `direct' studies of the general population or infer estimates.

Although it is reasonable to expect that daily variations in ambient particles contribute to (acute) respiratory symptoms in the general population, there is a shortage of suitable studies. In ExternE 1995 we used results from one study in California, USA: Krupnick *et al* (1990). This study gave very high values for symptom-day effects.

The PEACE study, conducted in a number of European countries, was designed to give direct estimates of air pollution effects on respiratory symptoms of school-children. Final papers from the PEACE study are not yet published; but we understand from preliminary communications that the results are generally negative. Against this background, we propose meantime *no* quantification of respiratory symptom effects in the general population.

Respiratory Symptoms in the General Population and Ozone
The quantification of ExternE 1995 was based on Krupnick *et al* (1990). We propose that these be used again, and we have scaled this upwards for 6-hr rather than for 1-hr ozone (Table 8.1).

8.5 Evaluation of Chronic Effects by Endpoint and Pollutant

Following review of the literature, we propose E-R functions *with particles only* for a limited number of chronic health endpoints (Table 8.1). In addition, some endpoints are included from prevalence studies in children of the effects of longer-term exposure (not of daily variations, specifically); though it seems that these refer to acute rather than to chronic conditions.

8.5.1 Chronic Mortality and Particles

There are several issues to be considered in relation to particles and chronic mortality.

Is there a causal relationship?

Mortality effects of long-term exposure to air pollution have frequently been studied using cross-sectional designs, for example in studies which seek relationships between mortality rates and air pollution concentrations in cities. 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, leading to doubt about their conclusions. Nevertheless, Lipfert (1994) 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 (Abbey *et al*, 1991; Dockery *et al*, 1993; Pope *et al*, 1995). In these studies the characteristics of the subjects (including where relevant their smoking habits) were collected on an individual basis. Adjustment for confounders at the individual level goes a long way to overcoming the methodological limitations of cross-sectional studies. The positive associations between PM and mortality found in two (Dockery *et al*, 1993; Pope *et al*, 1995) of these three key cohort studies are strong evidence of a real effect, though causality is neither proven nor universally accepted. Details of the two studies are summarised in Section 8.6.9.

Choice of study and E-R function for quantification

Of the two studies reporting positive effects, Pope *et al* (1995) is of much greater scale, involving about 300,000-550,000 individuals in 50-151 US metropolitan areas (for analyses of $PM_{2.5}$ and sulphates, respectively), compared with approximately 8,000 people in six cities studied by Dockery *et al* (1993). This difference in scale, with its associated reduced chances of confounding, is one reason why we have chosen to base risk estimates on Pope *et al* rather than on Dockery *et al*, despite the better pollution measurements in the latter. Another reason is that Pope *et al* give results that are intermediate between the higher estimates of Dockery *et al* and the lack of clear association in Abbey *et al* (1991).

Both studies suffer from the limitation that the estimated effects per $\mu g/m^3$ PM are based on relatively recent concentrations, and not on concentrations historically, which arguably are more relevant to the development of chronic disease and to associated premature death. The use of recent concentrations only will have led to over-estimates of the PM effect, insofar as PM concentrations have been reduced over time.

The effect of particles as estimated by Pope *et al* can be expressed in terms of $PM_{2.5}$ or of sulphates. These can in turn be re-expressed in terms of PM_{10} using the usual conversion factors, as for example in Dockery and Pope (1994). The E-R functions based on $PM_{2.5}$ and on sulphates have however different implications when converted to PM_{10} equivalent: the coefficient derived from $PM_{2.5}$ being about twice that derived from sulphates, expressed per $\mu g/m^3$ PM_{10}. We have chosen the E-R function based on $PM_{2.5}$, on the grounds of biological plausibility, with an estimated RR = 1.00386 per $\mu g/m^3$ PM_{10}.

Rescale the Pope* et al *(1995) functions, or not?

We considered scaling down the E-R functions from Pope *et al* (1995) to take into account of two factors. First, in considering chronic mortality, the biologically relevant exposures may have occurred some time ago; and in the US where the cohort studies were carried out, exposures historically to particulate air pollution will have been higher than in recent years. However Schwartz (1997) reported that on average across the USA, the reduction in annual average concentrations of TSP was not great through the 1970s and 1980s.

Secondly, there is the question of US-Europe transferability. Although it is likely that some scaling downwards of US functions is warranted, there is no basis in evidence for establishing what this should be, because there are no comparable European studies.

All things considered, we decided that, for the main analyses, the E-R functions from Pope *et al* (1995) would be used *without* any downwards scaling. If practicable, sensitivity analyses might be carried out where the same functions are used *but scaled downwards by a factor of two*.

Life table methods and years of life lost (YOLL)

The step from identifying an E-R function to estimating impacts is significantly more complex in the case of chronic mortality than it is for other health impact pathways, for two reasons. The simpler one is that an E-R function as identified via cohort studies is an estimate of the full pollution-related force of mortality, and so includes acute effects as well as effects of longer-term exposures. It is important to recognise this, in order to avoid double-counting. Secondly, a change in mortality hazard rates (age-specific death rates) in any one year affects not only mortality in that year, but the population-at-risk in future years; and so, even if age-specific death rates in later years were to revert to their earlier values, patterns of deaths in those later years would remain altered because the at-risk population had been changed.

Thus, it is necessary to track effects forward through time. This is done by the application of life table methods, a standard technique whose use in ExternE was developed and implemented at the IOM by Brian Miller, following a discussion initiated by Ari Rabl. The methodology is unnecessary when studying acute mortality effects only, on the understanding that on average the extra deaths identified by time series studies are deaths of vulnerable people, i.e. deaths brought forward by a relatively short length of time, probably less than a year on average. This is not so with chronic effects, where a mechanisms may be that of air pollution contributing to the development of chronic disease leading in due course to earlier death, but not until many years later.

The ExternE life table work develops and extends substantially some simple life-table calculations carried out by Bert Brunekreef on behalf of WHO and later published as Brunekreef (1998). The aim was to evaluate the effect on mortality (YOLL) of a 1-yr reduction of one $\mu g/m^3$ PM_{10}; i.e. with concentrations reverting to their original values after one year. This reversion is an unrealistic scenario in practice; its use in ExtrernE is to permit comparability with other effects, which are based on a 1-yr change in pollution concentrations. It implies however that effects need to be estimated only for the currently-alive population-at-risk; because the pollution change was for one year only, there are no effects on as yet unborn cohorts.

Calculations were based on:
a) population data, in 1-yr age-groups, separately for males and for females, up to age 94 inclusive, aggregated over the 12 EU countries at 1 January 1990, as published by Eurostat; and
b) age-specific death rates, again in 1-yr age-groups and separately for males and for females, up to age 100, from Germany as published by Statistiche Jahrbuch (1991).

(Sensitivity analyses from this and other studies showed that whereas the patterns of mortality clearly varied according to at-risk population and hazard rates, the estimated incremental effects of pollution were relatively insensitive to underlying population and death rates).

Effects were estimated separately for women and for men, and were restricted to those aged 30 years or more at exposure, i.e. the ages of the populations studied in the underlying cohort studies. The life table applications involved following the at-risk populations for 65 years or to age 100, whichever is earlier, under two sorts of scenario:
a) a baseline scenario assuming current hazard rates remain unchanged in future;
b) a changed scenario whereby hazards were modified to take account of a pollution change, and the effects followed through time even after the hazards had reverted to their original values.

Differences in results between i. and ii. were considered to be the pollution effect. This was summarised as life-years lost, scaled per 100,000 current live population, per 10 µg/m^3 PM_{10} reduction. Valuation was carried out before the results are summarised; i.e. economic valuation is linked with the detailed life table data, not with the summarised data. This enables adjustment of valuation by discounting for how far in the future the life-year is gained. The baseline VLYL was taken as 150,000 ECU. Discounting was applied at two rates: 3% and 11%; the 3% value was used.

In implementing the changed scenario (b., above), it was necessary to make some assumptions regarding latency; i.e. regarding the time-lag between a reduction in pollution and the associated percentage reduction in age-specific death rates. (The underlying cohort studies are uninformative about this.) The effects of different assumptions regarding latency were investigated. Briefly, estimated YOLL were relatively insensitive to different assumptions, but valuations were more sensitive, the more so when a higher discount rate was applied.

Results gave an estimated effect of a reduction of 720 YOLL per 100,000 current population (adults, aged 30 years or more) per 1-yr increase per µg/m^3 PM_{10}.

8.5.2 Chronic morbidity and particles

There is some direct evidence that long-term exposure to ambient particles contributes to the development of respiratory diseases in the general population (see, e.g. Lipfert, 1994;

COMEAP, 1995; US EPA, 1996). Also, if there really is an effect of long-term exposure on mortality, it is to be expected that this will be mediated via an effect on the development of cardio-respiratory disease. Thus, quantification of a PM effect on the development of chronic disease is desirable, if this can be done with sufficient reliability to be useful.

Adults

ExternE (1995) used some E-R functions from a study by Schwartz (1993). This was a cross-sectional study, though confounding factors had been measured and adjusted for at the individual level. A strength was that the data came from the (first) US National Health and Nutrition Examination Survey (NHANES I), and so covered a wide cross-section of US experience. A limitation was that risk estimates were expressed in terms of differences in prevalence following long-term exposure; and there is no easy way of deriving equivalent results on the effect of a 1-year pollution change, in terms of new cases of disease.

In the present study we have followed an approach proposed by Bart Ostro (see e.g. Rowe *et al*., 1994), and used results from a longitudinal study of Seventh Day Adventists (SDAs) in California (Euler *et al*., 1988; Abbey *et al*., 1991; Abbey *et al*., 1993; Abbey *et al*., 1995a, 1995b). Rowe *et al*. (1994) used risk estimates linking development of chronic bronchitis with long term exposure to TSP, as reported by Abbey *et al*. (1993); and obtained an E-R function in terms of PM_{10} by dividing the regression coefficient by 0.55, using a conversion factor from Dockery and Pope (1994). In the present project we have followed the same strategy, but have used have used an E-R function derived from one of the more recent papers from the SDA series (Abbey *et al*., 1995b), where the particle index was expressed in terms of PM_{10} rather than TSP.

There are important uncertainties regarding transferability: not just that the SDA studies were carried out in the USA, but also because of the distinctive lifestyle of those studied. Also, there are issues to be resolved in adapting the published E-R functions to take account of the effect of one year only of exposure to PM. The approach taken was to estimate, from Abbey *et al*., the increase in the number of new cases of chronic bronchitis *in one year*, associated with long-term average exposure; and treat this as equivalent to the number of new cases at any time in the future, associated with one year only of exposure. The actual study by Abbey *et al*. was based on new cases over a 10-year period; and so results were scaled to give new cases in one year only. Details are given in Section 8.6.10, later.

Again the question arises of whether these should be scaled down for application in Europe. Note however that we do not have any functions linking particles with the development of chronic *cardiovascular disease*; and, in the context of the evidence as a whole, this is an endpoint which we would expect is also affected. In view of this omission, and other considerations, we propose not to scale down the morbidity figures of Table 8.1.

These additional cases were valued as 'real' chronic illness; i.e. as a condition which lasts over time. This may have led to some over-estimation of effects, for two reasons. First, the definition of chronic bronchitis used by Abbey *et al*. was that of chronic cough *or* chronic phlegm; whereas it is conventional to require the joint occurrence of chronic cough *and* chronic phlegm as indicative of chronic bronchitis. Thus, the endpoint used was less severe than conventional chronic bronchitis (though many people with one of the two symptoms experience the other also) and so a lower valuation figure might be appropriate. Also, there is presumably some time-lag between biologically relevant exposure and effect; and so allowance should be made for some discounting. There is a lack of good information on what the time-lag might be.

Children

As in ExternE (1995), we use E-R functions linking PM with both bronchitis and chronic cough in children. These E-R functions are derived from Dockery *et al*, 1989, based on a cohort of over 5000 children, as part of the Harvard Six-Cities Study. The design of the Dockery *et al*. study was that of a conventional cross-sectional study, where health effects (from the previous two years) were related to concurrent (annual) average pollution concentrations, with adjustment for confounding factors at the individual level. Thus, the design examined health effects in relation to longer-term, not day-to-day, pollution characteristics; i.e. it was fundamentally different from the time series and panel studies which underlie most of the E-R functions for acute effects considered in this evaluation. Specifically, the pollution characteristics studied were indices of chronic, not acute exposure. The health outcomes, however, are best understood as additional *episodes* of illness, rather than as the development of a chronic condition. The valuations attached (see separate discussion) reflect this understanding.

8.6 Details of Specific Studies Underlying the Selected Exposure-Response Functions

8.6.1 Acute Mortality

Acute mortality and PM_{10}

Verhoeff *et al* (1996) considered PM_{10} air pollution and daily mortality in Amsterdam. Although Amsterdam was a city which took part in the APHEA (Air Pollution and Health; a European Approach) study, only the impacts on hospital admissions for respiratory diseases in Amsterdam and Rotterdam were reported (Schouten *et al*, 1996). It also only considered the pollutants of ozone, black smoke and SO_2, while PM_{10} was not measured. The APHEA meta-analysis for the PM_{10} impact on mortality (Katsouyanni *et al*, 1997) did not include Amsterdam and equated PM_{10} with PM_7 and PM_{13} for the other cities. Verhoeff *et al* (1996) has the advantage that conversion factors are not needed, as it reports associations of daily mortality directly with PM_{10}.

In the Verhoeff *et al* study (1996) daily mortality counts for Amsterdam (population 713,000 in 1992) were collected from the Municipal Population Register for the period 1986 to 1992. Air pollution levels as 24-hr averages for PM_{10}, SO_2, CO, O_3, and Black Smoke were collected over the same time period. Daily mortality counts were modelled using Poisson regression allowing for autocorrelation. There were no significant associations between mortality and SO_2 and CO. PM_{10} was significantly related to mortality with a RR = 1.062 (95% CI = 0.986, 1.144) for an increase of 100 μg/m^3 PM_{10}, adjusting for year of study, month, day of the week, epidemics of influenza-type illnesses, same-day temperature and relative humidity.

- This implies an increase in mortality of 0.0602% per µg/m^3 PM_{10}.

As part of the APHEA study Spix *et al* (1996) considered daily mortality and air pollution in Köln, Germany. Daily mortality and pollution levels were obtained for SO_2, NO_2, and PM_7, during the period 1975 to 1985. Adjustment in the Poisson regression modelling was made for long and short term trends, season, epidemics, weather, autocorrelation and overdispersion. SO_2 was significantly related to mortality, while particles measured as PM_7 was of borderline significance. PM_7 is assumed to be equivalent to PM_{10}. The regression coefficient was 0.0122 for an increase of 62 µg/m^3 of log transformed PM_{10}. Since the pollutant and mortality were both on the log scale the result can be expressed as a percentage increase in mortality for a percentage increase in pollutant using:

Percentage increase in mortality = (1 - r) x 100
where r = exp[β ln (proportionate increase in pollution)].

- As the regression coefficient was for an increase from 20 to 82 µg/m^3 PM_{10}, this represents an increase of 410% or 4.1. Hence r = exp (0.0122 x ln(4.1)) = 1.0174, that is the percentage increase in mortality = (1 - r) x 100 = 1.7% which is equivalent to 0.0278% per µg/m^3 PM_{10}.

The mean of the percentage increase in mortality from Spix *et al* (1996) and Verhoeff *et al* (1996) is thus 0.04% per µg/m^3 PM_{10}. This is consistent with the result of the APHEA meta-analysis (Katsouyanni *et al*, 1997) which reported a 2% increase in mortality per 50 µg/m^3 PM_{10}.

Acute Mortality and Ozone: Sunyer et al, 1996

Sunyer *et al* (1996) conducted a study in Barcelona relating air pollution and mortality, according to the APHEA protocol. Vehicle exhausts in Barcelona account for 35% of the total particulate level. Daily deaths for the period 1985 to 1991 were extracted from mortality registers for all cause and for respiratory and cardiovascular mortality. The pollutants measured were BS, ozone, SO_2 and NO_2. NO_2 and ozone were measured as 1-hr maximum daily values. Poisson regression modelling was applied to the daily counts of deaths and adjustment was made for temperature, relative humidity, with dummy variables for long and short time trends, and autocorrelation. Black smoke and SO_2 were significantly related to total mortality. In addition, ozone (1-hr max) and NO_2 (1-hr max) were both significantly related to total mortality, adjusted for year, season, day of the week, temperature, humidity, influenza and autocorrelation.

- For ozone the RR = 1.048 (95% CI = 1.012, 1.086) per 100 µg/m^3 O_3 (1-hr max) which is equivalent to 0.0469% per µg/m^3 O_3 (1-hr max). To convert to 6-hr mean, this is multiplied by 1.25 to give 0.0586% per µg/m^3 O_3 (6-hr mean).

Acute Mortality and SO_2: Touloumi* et al*, 1996 and Anderson* et al*, 1996

Touloumi *et al* (1996) studied air pollution and daily deaths in Athens for the period 1987 to 1991 as part of the APHEA study. One month, July 1987 was excluded as a climate 'outlier' as deaths increased by more than 100% due to an unusual heatwave. Pollutants measured included BS, SO_2 and CO. The daily death counts were analysed using Poisson regression modelling according to the APHEA protocol. Adjustments for seasonality, long term trends, temperature, humidity, day of the week and holidays and overdispersion were made in the regressions. In addition, the effects of BS and SO_2 were considered simultaneously. BS, SO_2 and CO were all significantly associated with all cause mortality. A stronger effect of SO_2 was seen on days when the levels of BS were greater than 100 µg/m^3.

- For SO_2 the percentage increase in mortality was 0.1101% per µg/m^3 SO_2 (24-hr mean).

Anderson *et al* (1996) also considered the relationship between SO_2 and daily mortality in London as part of the APHEA study. Daily counts of all cause mortality for all ages (excluding accidents) were collected for the period 1987 to 1992. Pollution levels were collected for ozone (1-hr max and 8-hr mean) from a single monitor in central London, and for NO_2 (1-hr mean and 24-hr mean) at the same site plus an additional monitor in wTJ4.519 ,(wTJ4.5

Barcelona gives an estimate of 0.0193% per µg/m^3 NO_2 (1-hr max) and converting to 24-hr mean using the factor of 1.67 gives percentage increase in mortality of 0.034% µg/m^3 NO_2 (24-hr mean). This function is only recommended for sensitivity analyses.

Acute Mortality and CO: Touloumi* et al*, 1994

Touloumi *et al* (1994) studied daily mortality data from Athens in the period 1984 to 1988. Air pollution measures used were black smoke, sulphur dioxide and carbon monoxide. Counts of deaths in July, 1987 were excluded as deaths doubled in this time due to an unusual heatwave. In the regression analysis autoregressive models with log-transformed daily mortality as the dependent variable were used, to allow for serial correlation of mortality data. All models were also adjusted for temperature and relative humidity (both lagged 1 day), as well as year, season, and day of the week.

The mean daily CO level was 5.76 mg/m^3 over the full time period. As 1 mg/m^3 is equivalent to 0.873 ppm CO, the mean daily CO was 5.028 ppm.

There was a highly significant association between daily mortality and daily CO, adjusting for temperature, relative humidity, year, season, and day of the week with regression coefficient 0.060 (SE = 0.010). As both mortality and CO were related on the log scale the regression coefficient can be re-expressed in proportionate terms:

Percentage increase in mortality = (1 - r) x 100

where r = exp [β ln (proportionate increase in pollution)].

- Hence a 10% increase in CO gives r = exp (0.06 x ln(1.10)) = 1.0057 and so is associated with a 0.57% increase in mortality with lower and upper estimates of 0.48% and 0.67% representing one standard error. As the daily mean CO level was 5.76 mg/m^3, a 10% increase implies a 0.97% increase in daily mortality per mg/m^3 CO. Converting this result to 24-hr mean by multiplying by 1.5 gives 1.45% per mg/m^3 CO (24-hr mean) or 0.00145% per µg/m^3 CO (24-hr mean).

Note that although CO was significantly related to mortality, after adjusting for SO_2 and/or Black Smoke, this relationship became non-significant. The above function for CO is recommended for sensitivity analyses only.

8.6.2 Hospital Admissions

Respiratory Hospital Admissions and PM_{10}: Dab* et al*, 1996

Dab *et al* (1996) considered the association between daily respiratory deaths and respiratory hospital admissions in Paris over the period 1987 to 1992 as part of the APHEA study. Daily pollution levels for BS, PM_{13}, SO_2, NO_2 and ozone were collected in the same time period. As in all the APHEA studies Poisson regression modelling was used in the analysis. Daily mortality from respiratory causes was significantly related to both PM_{13} and BS. Both were also significantly associated with respiratory hospital admissions.

For respiratory hospital admissions, the RR = 1.045 for 100 µg/m^3 PM_{10} which is equivalent to 0.0440% per µg/m^3 PM_{10}, assuming PM_{10} is equivalent to PM_{13}.

The mean admission rate for respiratory conditions was 79 per day from a population of 6.14 million giving a daily rate of 1.2866 per 100,000. Applying the percentage increase above gives an increase in respiratory admissions of 0.207 per 100,000 per day per year per µg/m^3 PM_{10}.

Respiratory Hospital Admissions and O_3, SO_2 and NO_2: Ponce de Leon* et al*, 1996

Ponce de Leon *et al* (1996) considered the relationship between air pollution and daily hospital admissions for respiratory disease in London for the periods 1987 to 1988 and 1991 to 1992. Emergency admissions for respiratory diagnoses (ICD 460 - 519) were obtained from the Hospital Episode system for London health districts. Pollution levels measured were BS, SO_2 (24-hr mean), NO_2 (1-hr max and 24-hr mean) and ozone (1-hr max and 24-hr mean). Ozone was significantly related to daily respiratory admissions. SO_2 showed a positive relationship to daily admissions but this relationship was less consistent than that for ozone. NO_2 was significantly related to all age respiratory admissions.

The mean number of daily admissions for respiratory causes was 125.7 for a population of 7.2 million which is equivalent to 1.7458 per 100,000 per day.

- For ozone the increase in admissions for respiratory conditions had a RR = 1.0293 per 26 ppb (8-hr O_3) which is equivalent to 0.111% per ppb (8-hr O_3). It is assumed that 8-hr is equivalent to 6-hr mean and dividing by 1.997 to convert to µg/m^3 gives 0.0556% per µg/m^3 O_3 (6-hr mean). Using the daily admission rate above and the percentage increase gives an increase in respiratory admissions of 0.354 per 100,000 per year per µg/m^3 O_3 (6-hr mean).

- For SO_2 the increase in respiratory admissions was given by RR = 1.0092 per 29 µg/m^3 SO_2 (24-hr mean) which equivalent to 0.0316% per µg/m^3 SO_2 (24-hr mean). Using the daily admission rate above gives an increase of respiratory admissions of 0.204 per 100,000 per year per µg/m^3 SO_2 (24-hr mean).

- For NO_2 the increase in respiratory admissions has a RR = 1.0114 per 27 ppb NO_2 (24-hr mean) which is equivalent to 0.0420% per ppb NO_2 (24-hr mean). Dividing by 1.913 to convert to µg/m^3 gives an increase of 0.0220% per µg/m^3 NO_2 (24-hr mean). Relating this to the mean admission rate per day gives 0.140 per 100,000 per year per µg/m^3 NO_2 (24-hr mean).

Cardiovascular hospital admissions and PM_{10}, CO: Schwartz and Morris, 1995

Schwartz and Morris (1995) examined the relationship between air pollution and cardiovascular admissions in Detroit, Michigan for people aged 65 years and older during the period from 1986 to 1989. Air pollutants considered were PM_{10}, ozone, SO_2 and CO. CO was measured as it is a known risk factor for cardio-vascular disease. Daily admissions were obtained for (ICD 9th revision), ischaemic heart disease (410 - 414), cardiac dysrhythmias (427), and congestive heart failure (428) in the population aged 65 years and older (517,000).

This data was obtained from standardised reports containing date of admission and code for discharge diagnosis (ICD-9).

For each day PM_{10} levels were extracted for all monitoring stations in Detroit and averaged. Ozone was obtained for 85% of the days as it was discontinued when ozone levels were very low. Data for CO and SO_2 were available for every day of the study period.

Poisson regression was used to model daily admissions. Dummy variables for each month (48) were used to model long term fluctuations and so remove seasonal, between month and yearly patterns. Linear and quadratic terms were also added to include long term temporal trends in admissions. Eight categories of temperature and dew point were added to model the weather and potential non-linearities. In addition, dummy variables for each day of the week were included. Nonparametric smoothing was used to assess patterns in the residuals which could indicate poor fit in time periods, or in temperature or humidity ranges. Pollution variables were only added after adequate control for time trends and weather. The presence of autoregression was tested by examining previous days admissions up to a lag of three days, and included if significant.

In addition to the above modelling strategy, sensitivity of the modelling was assessed in a number of ways:

1. robust regression methods used to assess impact of outliers;
2. the weather model was specified deleting all days less than the 5th percentile and greater than the 95th percentile of temperature;
3. dummy variables added for eight categories of temperature on each of previous 2 days to assess possible temperature lags;
4. Generalized additive modelling (GAM) used to assess interactions of temperature and humidity;
5. repeat regressions restricting sample to days below current air quality standards for PM_{10}.

Pollutants were added as same day pollution and lags of up to two days. Pollutants were initially examined separately and then two-pollutant models were considered to assess their independent contributions to explaining hospital admissions.

The E-R functions described below for congestive heart failure are to be used in the main implementation while those for ischaemic heart disease are only for sensitivity analyses.

Congestive heart failure (CHF) and PM_{10}

There was a significant relationship between admissions for congestive heart failure and PM_{10} with a RR = 1.032 (95% CI = 1.012, 1.052) for an increase of 32 µg/m^3 PM_{10}. As Poisson regression was used this can be interpreted as a 0.0985% increase per µg/m^3 PM_{10}. The mean daily admission rate for CHF was 26.6 for a population of 517,000 aged 65 years and older. Combining this with the percentage increase above gives an increase in admissions for CHF = 0.000985 x (26.6/517,000) = 5.068 x 10-8 per µg/m^3 PM_{10} per day. Multiplying by 365 to obtain the annual increase gives:

- Increase in annual admissions (aged 65+) for CHF = 1.85 per 100,000 per µg/m^3 PM_{10}.

Congestive heart failure (CHF) and CO

There was also a significant association of CO with admissions for congestive heart failure with a relative risk = 1.022 with 95% confidence interval (1.011, 1.033) for an increase of 1.28 ppm CO (the interquartile range). On adding PM_{10} to this model the association with CO remained significant and hence suggests an independent effect of CO. As the Poisson model was used this can be interpreted as the percentage increase in CHF admissions per increase in CO equal to 1.72% per ppm CO (1-hr max).

The daily mean hospitalisation rate for CHF was 26.6 in this study per 517,000 per day and using this value gives an annual increase of 0.0172x26.6x365/5.17 = 32.3 per 100,000 per ppm CO (1-hr max). To convert to 24-hr metric this result was multiplied by 2 and to convert to mg/m^3 CO was divided by 1.165 to give:

- Increase in annual admissions for CHF = 55.5 per 100,000 per mg/m^3 CO (24-hr mean)

- or increase in annual admissions (aged 65+) for CHF = 5.5 x 10-7 per µg/m^3 CO (24-hr mean).

Ischaemic Heart Disease (IHD) and PM_{10}

- There was a significant relationship between PM_{10} and admissions for IHD with a RR = 1.018 (95% CI = 1.005, 1.032) for an increase of 32 µg/m^3 PM_{10}. This implies a percentage increase of 0.0562% per µg/m^3 PM_{10}. Using the daily mean rate of admissions of 44.1 for 517,000 (aged 65+) gives an increase of 0.000562x44.1x365/517,000 = 1.75 per 100,000 per µg/m^3 PM_{10}.

Ischaemic Heart Disease (IHD) and CO

There was also a significant association of CO with admissions for ischaemic heart disease with a relative risk = 1.010 with 95% confidence interval (1.001, 1.018) for an increase of 1.28 ppm CO (the interquartile range). On adding PM_{10} to this model the association with CO became non-significant. Thus the percentage increases in IHD admissions was 0.781% per ppm CO (1-hr max).

The mean hospitalisation rate for IHD was 44.1 in this study per 517,000 per day and using this value gives 0.00781x44.1x365/517,000 = 24.3 per 100,000 per ppm CO (1hr max). To convert to 24-hr metric this result was multiplied by 2 and to convert to mg/m^3 CO was divided by 1.165 to give:

- Increase in annual admissions for IHD = 41.7 per 100,000 per mg/m^3 CO (24-hr mean)

- or increase in annual admissions (aged 65+) for IHD = 4.17 x 10-7 per µg/m^3 CO (24-hr mean).

Cerebrovascular Hospital Admissions and PM_{10}:: Wordley* et al*, in press

Wordley *et al* (in press) considered the relationship between hospital admissions and daily pollutant levels in Birmingham, UK. The study concentrated on the health impacts of PM_{10}.

Daily admissions for asthma, bronchitis, pneumonia, COPD, acute IHD, acute cerebrovascular disease, all respiratory conditions and all circulatory conditions were obtained from the West Midlands Regional Health Authority.

Multiple linear regression was used to assess the association adjusting for day of the week, month, linear trend, relative humidity and temperature. Poisson regression modelling was also carried out as a check on the assumption of normality for the distribution of deaths. Relative risks of admissions were calculated by comparing the risk of admissions over the threshold with mean risk of admissions over the whole period.

The daily mean admissions for CVA was 6.6 for a population of one million which is 0.66 per 100,000.

Using the normal model the increase in admissions for CVA was 0.0137 per million per $\mu g/m^3$ PM_{10} which is equivalent to 0.500 per 100,000 per year per $\mu g/m^3$ PM_{10}. The Poisson model gave a percentage increase of 0.21% per $\mu g/m^3$ PM_{10}. Combining this with the daily mean admission rate of 0.66 per 100,000 gave an increase in admissions for CVA of 0.504 per 100,000 per $\mu g/m^3$ PM_{10}. The similarity of this result to that obtained assuming a normal distribution for admissions justifies the normal assumption in the modelling.

8.6.3 Emergency Room Visits

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 made for ExternE (1995) 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%) 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) x 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 x 4398)/(25 x 14) = 0.72 per 100,000 per µg/m^3 PM_{10}.

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 an estimate of 0.367% per µg/m^3 PM_{10}.

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;

- Increase in ERV for asthma per 100,000 per year = 1.29 per µg/m^3 PM_{10}.

For transfer from the US to Europe this value is reduced by one half to give:

- Increase in ERV for asthma per 100,000 per year = 0.645 per µg/m^3 PM_{10}.

PM_{10} and ERVs for croup in pre-school children: Schwartz* et al*, 1991

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 laryngotracheitis, 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 NOx 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 NOx 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 NOx 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 E-R relation:

- Additional cases of childhood croup per 100,000 per year = 1.6 per µg/m^3 TSP.

Using the conversion factors of Dockery and Pope (1994) (which of course may apply only approximately to Germany), we divide this estimate by 0.55 to give:
Additional cases of childhood croup per 100,000 per year = 2.91 per µg/m^3 PM_{10}.

Ozone and ERVs for asthma: Cody* et al*, 1992 and Bates* et al*, 1990.

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 SO_2 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. PM_{10} 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 O_3 (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 O_3, in all analyses. The sparse PM_{10} data showed no additional effect. (Ozone was unrelated to ERVs for bronchitis, where a weak association with PM_{10} was found.) Omitting weekends, or adjusting for autocorrelation, made little difference to the results.

Detailed results varied somewhatfit (f94(e[(De47)-1.5e47)-1.5e47 Tm42l7(o)0.c)5(sc)5(sc)t8246 Tw[e55e4

8.6.4 Restricted Activity Days (RADs)

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 Work Loss Days (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, from 68 metropolitan areas.

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. Adjustments were made for between-city differences in factors such as time spent out of doors, building construction, and health practices by using a fixed effects model; i.e. by focusing the analysis on the deviation of individual observations from their city means. 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 (daily) minimum temperature in the two week period of recall for each individual. A variable was also added to indicate whether the individual was working or not.

The separate year-by-year analyses for the six years 1976 to 1981 gave six coefficients, each of which was positive and highly significant statistically ($p<0.01$). The estimated size ranged from 0.00284 to 0.00900, with a geometric mean regression coefficient of 0.00438 per µg/m^3 FP. (Ostro, 1987, reports coefficients scaled upwards by a factor of 100.) 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. Averaging across years, and using the conversion factor (FP=) $PM_{2.5}$ = 0.6 PM_{10} (Dockery and Pope, 1994), this implies a percentage change of 0.263% per µg/m^3 PM_{10}.

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 percentage increase of 0.263% per µg/m^3 PM_{10} gives the following annual increase:

- Increase in RADs per 1,000 adults per year = 49.9 per µg/m^3 PM_{10}

For transfer from US to Europe the E-R function is reduced by one half to give:

- Increase in RADs per 1,000 adults per year = 25 per µg/m^3 PM_{10}.

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 $\mu g/m^3$) 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 0.00085 (SE = 0.002) per $\mu g/m^3$ of ozone. This is equivalent to 0.00200 (SE = 0.00470) per ppb ozone. (1 ppb = 1.997 $\mu g/m^3$ O_3).

As a Poisson regression model was used the regression coefficients can be interpreted as a percentage increase per ppb ozone. Thus, the percentage increase in MRADs is 0.200% per ppb ozone.

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 result:

- Increase in MRADs per 1,000 population per year = 15.6 per ppb ozone.

To convert to $\mu g/m^3$ and the 6-hr mean metric this estimate is divided by 1.997 and multiplied by 1.25 to give:

- Increase in MRADs per 1000 population per year = 9.76 per $\mu g/m^3$ ozone (6-hr mean).

8.6.5 Provocation or exacerbation of asthma in children

Bronchodilator usage in asthmatics: Roemer* et al*, 1993

Roemer *et al* (1993) studied a panel of 73 children (mean age 9.3 yrs) in the two Dutch cities of Wageningen and Bennekom over the winter from December 1990 to March 1991, a total of 79 days. The study panel consisted of those children who gave a positive response to one or more of the following three questions from a WHO questionnaire:

- 'Did your child have attacks of shortness of breath while wheezing during the past year?';
- 'Does your child cough like this (i.e. cough during day or night or most days in autumn and winter) for 3 months a year?'; and
- 'Has your child been treated for asthma by a specialist during the last year?'.

Of the 1253 responders 131 gave a positive response to at least one of the three questions. From this group a random sample of 74 was selected. One child dropped out leaving 73 for the panel study, of which 71% (52) had attacks of shortness of breath with wheeze over the last year, while 21% (15) were treated for asthma by a specialist.

On this basis, we think that the study is best considered as a study of asthmatic children.

There was a significant association between the prevalence of same day bronchodilator usage and measures of daily PM_{10}, SO_2 and BS. The odds ratio (RR) was 1.023 (95% C.I. = 1.007, 1.039) for an increase in PM_{10} of 10 $\mu g/m^3$. This is equivalent to a percentage increase of 0.23% per $\mu g/m^3$ PM_{10} on the *odds* of bronchodilator usage on any day.

The background bronchodilator use during the period of this winter-time study was 10.3%, equivalent to an odds of 0.1148, i.e. (0.103)/ (0.897). Applying the RR of 1.0023 per $\mu g/m^3$ PM_{10} to this background odds gives a new odds of 0.11509 which, after conversion to probability or prevalence results in an:

- Increase in bronchodilator use per 1000 child asthmatics per *day* = 0.2133 per $\mu g/m^3$ PM_{10}.

We annualise these figures to give

- Increase in bronchodilator use per 1000 child asthmatics per *year* = 77.9 per $\mu g/m^3$ PM_{10}.

Note that this annualisation involves two extrapolations beyond the available data, either of which might be questioned:

a) An assumption that the PM-related increase observed in this winter-time study applies throughout the year; and
b) An assumption that the winter-time background prevalence of 10.3% bronchodilator usage also applies throughout the year.

Lower respiratory symptoms (wheeze) in asthmatics: Roemer* et al*, 1993

In the same study of asthmatic children (Roemer *et al* 1993), the symptom of wheezing was also recorded by the participating children in the two Dutch cities. The mean daily prevalence of wheeze was 9.6% over the (winter-time) period of the study.

There was a significant association between the prevalence of wheeze and PM_{10}, SO_2 and BS. The RR was 1.033 (95% C.I. = 1.013, 1.054) for an increase in PM_{10} of 10 $\mu g/m^3$. This is equivalent to a percentage increase of 0.325% per $\mu g/m^3$ PM_{10} on the odds of reporting wheeze on a given (winter-time) day.

Relating this percentage increase to the background odds of 0.106195, converting back to probabilities (prevalences), and annualising, gives:

- Increase in wheeze per 1000 children asthmatics per year = 102.9 per $\mu g/m^3$ PM_{10}.

The annualisation involves similar extrapolations to those for bronchodilator usage, above.

Cough in asthmatics: Pope and Dockery, 1992

Pope and Dockery (1992) studied a panel of symptomatic and asymptomatic children (n = 39) in Utah Valley in the winter of 1990-1991. There were significant associations between respiratory symptoms, especially cough in both samples, and same day PM_{10}.

In the symptomatic sample, the logistic regression linking daily cough with PM_{10} was 0.506 (se = 0.143) for an increase of 100 µg/m^3 PM_{10}, equivalent to the percentage increase of 0.507% per µg/m^3 PM_{10}.

The mean daily prevalence of cough was 17.5%, equivalent to an odds of 0.21212. Relating the odds ratio (RR) of 1.00507 per µg/m^3 PM_{10} to this background odds, converting back to probabilities (prevalences), annualising, *and reducing the result by one half for transfer from the US to Europe* gives:

Lower respiratory symptoms (wheeze) in asthmatics: Dusseldorp* et al*, 1995

In the same study (Dusseldorp *et al* 1995), the panel of 32 symptomatic adults recorded lower respiratory symptoms such as wheeze for 67 days over the same winter-time period between October 11 and December 22, 1993. The mean daily prevalence of wheeze was 8.1%.

There was an association between the prevalence of same day wheeze and PM_{10}. The RR was 1.25 (95% C.I. = 0.79, 1.97) for an increase in PM_{10} of 100 µg/m^3, equivalent to a percentage increase of 0.223% per µg/m^3 PM_{10}.

Relating this percentage increase to the background odds of 0.08814, converting back to probabilities (prevalences), and annualising, gives:

- Increase in wheeze per 1000 adult asthmatics per year = 60.6 per µg/m^3 PM_{10}.

The usual caveats about extrapolation apply.

Cough in asthmatics: Dusseldorp* et al*, 1995

Also in the same study, Dusseldorp *et al* (1995) studied daily cough in the same panel over the same winter-time period. The mean daily prevalence of cough was 18.4%.

There was an association between the prevalence of same day cough and PM_{10}. The RR was 1.31 (95% C.I. = 0.97, 1.76) for an increase in PM_{10} of 100 µg/m^3, equivalent to a percentage increase of 0.306% per µg/m^3 PM_{10}.

Relating this percentage increase to the background odds of 0.225, converting back to probabilities (prevalences), and extrapolating as before to annualise, gives:

- Increase in cough per 1000 adult asthmatics per year = 167.6 per µg/m^3 PM_{10}.

8.6.7 Ozone and 'Asthma Attacks': Whittemore and Korn, 1980

Data from the diaries of 16 panels of asthmatics living in the Los Angeles area, California were collected for 34 weeks in the period between 1972 and 1975 for the US Environmental Protection Agency. The areas included Santa Monica, Anaheim, Glendora, Thousand Oaks, Garden Grove and Covina with a total of 443 asthmatics in the study.

Panellists had to have experienced at least one attack in the year before the study, had a history of wheezing and dyspnea with each attack and lived within two miles of the community's monitoring station. Questionable asthma diagnoses were verified by a physician. Information was collected on demographic variables, smoking habits, occupational exposures and other factors apparently related to their attacks.

Attacks were recorded in a weekly diary, as well as information on medication use which was then returned at the end of the week. As there was over reporting of attacks during the first two weeks of the study, these weeks were excluded from the analyses.

Ozone levels as daily one hour maximum in ppm were collected by country air pollution control districts. Total suspended particulates (TSP) were also collected and used in the analyses. Weather indicators used in the analyses were minimum temperature, relative humidity, and average wind speed.

Analysis of the panel data was based on a multiple logistic model of each individual's attack probability. Variables included in the logisitic model were daily attack data, the presence or absence of an attack the previous day, daily levels of oxidant, TSP, minimum temperature, relative humidity, average wind speed, time since start of study and day of the week. An advantage of this approach is that each individual has a regression coefficient for each variable specific to them and susceptible individuals can easily be identified by their large regression coefficients. Days with missing attack data or missing pollutant data were excluded. In order to obtain summary estimates the individual coefficients were combined over the panel using both fixed effects and random effects models. The fixed effects model assumes the effect of the variable does not vary from person to person while the random effects model allows the coefficients to vary.

There was a statistically significant relationship between oxidant level and the probability of an asthma attack. TSP was also significantly related to the probability of an attack. The presence of attack on the previous day, day of the week, and day of study were highly significant predictors of an attack. From the fixed effects model the regression coefficient for oxidant was 1.66×10^{-3} per ppb oxidant (1-hr max). Note that we assume oxidant is equivalent to ozone. This estimate is multiplied by 1.25 to convert to the metric of 6-hr mean and divided by 1.997 to convert to $\mu g/m^3$ giving 1.039×10^{-3} per $\mu g/m^3$ ozone (6-hr mean). Assuming the mean daily attack rate of 13% per asthmatic (i.e. an average of 0.13 attacks per day per asthmatic) and applying this to the regression coefficient above gives an increase of 0.000117 attacks per day per asthmatic and annualising we obtain:

- Annual increase of 0.0429 asthma attacks per asthmatic per $\mu g/m^3$ ozone (6-hr mean).

8.6.8 Respiratory symptoms in the general population

Ozone and symptom days: Krupnick* et al*, 1990

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 E-R 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, NO_2, SO_2) 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.

Results for ozone were taken from this paper where the ozone effects are adjusted for particles. The key relationship is summarised by the regression coefficient 0.0055 (SE = 0.0027) per O_3 (pphm), which is equivalent to 0.00055 (SE = 0.00027) per O_3 (ppb).

The results were not applied to children as no significant relationship was found for children. 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.

The baseline probability of any (adult) subject reporting any symptom is given as 0.19. This is therefore the stationary probability described also as $P1 = p0/ (1 - p1 + p0)$ where

p1 = mean transition probability (illness day t given illness on day t-1) and
p0 = mean transition probability (illness on day t given no illness on day t-1).

The mean values of p0 and p1 were not stated in the paper. However, knowledge that the baseline (stationary) probability is 0.19 overall, and that the log odds of p1 relative to p0 is 4.065, allows these to be estimated as p0 = 0.0550 and p1 = 0.7655. (Rowe *et al* (1994), report a personal communication with Krupnick *et al* (1990), that the mean adult values were p1 = 0.7775 and p0 = 0.0468; i.e. the estimates we used were similar.)

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

$$\pi'_1 = \beta \frac{p_0(1 - p_1)(1 - p_0 + p_1)}{(1 - p_1 + p_0)^2}$$

where, after substitution for p_0 and p_1, $\lambda = 0.2632$.

Thus the health effect of ozone is given by 0.00055 x 0.2632 = 0.0001448 and so the annual increase in symptoms per 1000 adults = 52.8 per ppb O_3 (1-hr max). To convert to µg/m^3 this was divided by 1.997 and multiplied by 1.25 to convert to 6-hr mean giving the final result:

- Annual increase in symptom days per 1000 adults at risk = 33.0 per µg/m^3 (ozone 6-hr mean).

8.6.9 Chronic Mortality

Dockery* et al, *1993

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.

Pope* et al, *1995

Pope *et al* (1995) examined mortality and ambient particles in a very large prospective study of 552,138 individuals, followed-up over eight years as part of a wider study of the American Cancer Society (ACS). These individuals lived in 151 US metropolitan areas for which ambient concentrations of sulphates at the start of the follow-up were available. Recent concentrations of fine particles ($PM_{2.5}$) were available for 50 metropolitan areas, including almost 300,000 of the overall cohort. The mean sulphate level was 11.0 µg/m^3 (sd 3.6; range

3.6-23.5 µg/m^3) while the mean $PM_{2.5}$ level was 18.2 µg/m^3 (sd 5.1; range 9.0 to 33.5 µg/m^3). The two measures of ambient particles were highly correlated (r = 0.73). There were no estimates of historical exposures.

Adjusted mortality risk ratios were estimated using regression analyses based on the Cox proportional hazards model. In effect, these analyses compared across metropolitan areas the age-specific death rates with pollution concentrations, after adjustment for a wide range of confounding factors (age, sex, race, cigarette smoking, exposure to passive cigarette smoke, body mass index, drinks per day of alcohol, education and occupational exposure) at the individual level. Results are presented as the percentage increase in mortality hazards (age-specific death rates) between the highest and lowest polluted areas. It was assumed that the same percentage increase applied at all ages.

Results showed that both indices of particles were statistically significantly associated with all cause mortality hazards. The RR for sulphates was estimated at 1.15 (95% CI = 1.09, 1.22) for an increase of 19.9 µg/m^3 SO_4 (i.e. the difference between the highest and lowest polluted areas). The estimated RR for fine particles was 1.17 (95% CI = 1.09, 1.26) for an increase of 24.5 µg/m^3 $PM_{2.5}$.

The result for $PM_{2.5}$ of RR = 1.17 (1.09, 1.26) is equivalent to an increase of 40.8 µg/m^3 of PM_{10}, using the usual conversion factor that $PM_{2.5}$ is approximately 0.6 PM_{10} (Dockery and Pope, 1994). This implies RR = 1.00386 per µg/m^3 PM_{10} or:

- Percentage increase in chronic mortality = 0.386% per µg/m^3 PM_{10}.

8.6.10 Chronic Morbidity in Adults

The function proposed for chronic morbidity in adults derived from the most recent (Abbey *et al* 1995a, 1995b) of a longer series of papers (e.g. Euler *et al* 1988, Abbey *et al* 1991, Abbey *et al* 1993,), which describe the design, conduct and analysis of a longitudinal study of air pollution and health in a cohort of Seventh-Day Adventists (SDAs) in California, USA.

Characteristics of the SDA population

The original cohort enrolled in April 1997 for the air pollution study consisted 6340 white, non-Hispanic adult members of the Seventh-Day Adventist church who had lived for the last 10 years within five miles of their current residence (Abbey *et al.*, 1995b). The SDA church has rules against smoking; and this cohort therefore provided a possibly unique opportunity to study the effects of air pollution with little or no confounding from tobacco smoking.

Results used for chronic morbidity in the present project are based on a sub-cohort (the 'Respiratory Symptoms Cohort') of 3914 adults who completed a standardised respiratory symptoms questionnaire both in April 1997 and April 1998. The symptoms questionnaire was designed to obtain information about self reported symptoms of chronic respiratory disease, together with confounding factors such as occupational exposures, hours of freeway driving, time spent indoors, detailed smoking histories, and environmental tobacco smoke exposure.

Age at enrolment of the study group ranged from 27 to 95 years (mean age 56y); almost two-thirds (64%) were female. Based on responses to the symptoms questionnaires, any individual who smoked between 1977 and 1987 was excluded from the study. However, among the participants in the cohort were 15% who had smoked before 1977, who had joined the SDA church later in life. In addition, 30% of the cohort had lived with a smoker, and 42% had worked with a smoker. Concerning occupational exposure, 12% had experienced exposure to dust, and 15% had been exposed to fumes.

Estimation of long-term exposures; exposure indices used in the analyses

Monthly residence and work location histories were obtained from 1967 to 1987 for participants in the study. Regression equations relating PM_{10} to TSP levels were constructed for this 20 year time period. These, along with residence and work information, were used to construct mean monthly concentrations of PM_{10} for participants in the study.

In the regression analyses, the incidence of new cases of disease was related to PM_{10}, as the mean concentration in $\mu g/m^3$, as well as the number of hours per year exceeding various levels of PM_{10}. In the latter analysis the cut-off points used were 40, 50, 60, 80, and 100 $\mu g/m^3$. This form of analysis gives results from which it is difficult to extract dose-response relationships, but they do provide useful information on possible thresholds. The same set of covariates were used in all regression analyses.

Health endpoints

The respiratory symptoms questionnaire used was based on well-validated sources: the British Medical Research Council, adapted for use in the USA by the United States National Heart and Lung Institute (NHLI). The 1987 version included additional questions based on the American Thoracic Society questionnaire.

Algorithms, designed to require clinical significance of symptoms in terms of persistence and severity, were used to classify individuals as having chronic bronchitis, asthma or emphysema on either or both of the two occasions (April 1977 and April 1987). These algorithms were based on 21 respiratory symptoms questions. Specifically, that for chronic bronchitis included chronic cough and/or sputum production on most days, for at least three months/year, for at least two years. In many studies, *both* chronic cough and chronic sputum production are required for the presence of chronic bronchitis; the present study's inclusion of either indicates a less severe condition. A diagnosis of definite asthma required a history of wheezing and a physician diagnosis. Definite emphysema was assumed if a participant was told by a physician that he had emphysema and if he experienced shortness of breath during exercise or when walking.

Definite airways obstructive disease (AOD) was defined as a classification of at least one of definite chronic bronchitis or definite asthma or definite emphysema. New cases of AOD were individuals who had definite symptoms in 1987, but not in 1977. Those with definite symptoms in 1977 were excluded from the analysis. Individuals whose responses indicated the presence of possible symptoms in 1977 were identified but remained in the data set.

PM_{10} and chronic bronchitis: Abbey* et al*, 1995b

Regression analyses reported by Abbey *et al* (1995b) showed a statistically significant relationship between the development of new cases of definite symptoms of chronic bronchitis and estimates of the long term ambient PM_{10} concentrations. In these analyses, adjustment was made for years smoked, years lived with a smoker, years worked with a smoker, possible symptoms in 1977, age, gender and education.

The relative risk for chronic bronchitis was 1.15 for mean concentration of PM_{10} of 20 $\mu g/m^3$, which was identical to that for AOD. A confidence interval was not given, but in the analysis of the same outcome with PM_{10} exceedance frequency of 42 days per year above 100 $\mu g/m^3$ a confidence interval was given thus; RR = 1.17 (95% C.I. = 1.01, 1.35), and so from this the standard error for the regression coefficient was taken as 0.07352. We have used this standard error in place of the unreported standard error associated with the effect of 20 $\mu g/m^3$ mean concentration of PM_{10}.

As logistic regression was used in the analysis, the increase in log (odds) was 0.006988 per $\mu g/m^3$ PM_{10} which corresponds to a percentage increase of 0.70% per $\mu g/m^3$ PM_{10}.

The proportion of new cases over the 10 years of follow-up given in this study (Abbey *et al* 1995b), was 234/3310; i.e. the probability of a new occurrence of chronic bronchitis was 0.007069 per year. Converting this background rate to an odds, and linking with the regression coefficients as above, we find:

- Annual increase in new cases of chronic bronchitis per 100,000 adults (aged 27+) = 4.9 per $\mu g/m^3$ PM_{10}.

Some comments on alternative values from the SDA studies

As noted above, a dummy variable indicating possible symptoms in 1977 was included by Abbey *et al*. (1995b) in the regression analysis. Presence of symptoms at the start of the 10-year period may have been in part the result of earlier exposure to air pollution; and so it is likely that adjustment for presence of symptoms in 1997 will have led to an underestimate of the overall effect of ambient particles. Indeed, sensitivity analyses confirmed this: without adjustment for symptoms in 1977, the size and statistical significance of the regression coefficient increased. However, the coefficient after adjustment is the more appropriate for effects which occur with a short time-lag after exposure; i.e. for use without discounting.

A further analysis considered an adjustment for time spent indoors. A factor of 0.7 was estimated and applied to each individual estimated monthly mean concentration according to the estimated time spent indoors that month. The statistical significance of the mean PM_{10} concentration did not change in the regression models with exposure adjusted in this way, but the size of the regression coefficient again increased.

Rowe *et al* (1994) used risk estimates, linking development of chronic bronchitis with long term exposure to TSP from the same study but as reported in an earlier paper (Abbey *et al* 1993), to obtain an exposure function for PM_{10} by dividing the regression coefficient by 0.55. The results were similar to the above calculations with estimates of new cases of chronic bronchitis per 100,000 adults aged 27 yrs and over of 3.0 (low), 6.1 (mid), and 8.1 (high) per average annual increase of PM_{10} ($\mu g/m^3$).

It is possible to derive E-R functions expressed in terms of the more inclusive endpoint of AOD rather than chronic bronchitis. It seems that results would be similar, regardless of which one of the two is used. For example, there were 234 new cases of chronic bronchitis and 272 new cases of AOD in the entire (sub-)cohort; i.e. of the three causes (definite chronic bronchitis, definite asthma and definite emphysema) contributing to AOD, chronic bronchitis was dominant. Also, as noted above, the RR for chronic bronchitis and for AOD were the same. The estimated RR for asthma was higher but not quite significant statistically at the 5% level. However, a recent comprehensive review (COMEAP, 1995) found no good evidence across studies generally linking ambient air pollution and development of asthma. This influenced our choice of the more specific chronic bronchitis endpoint.

8.6.11 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

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 = 2.5, 95% CI 1.1, 6.1) and to chronic cough (OR = 3.7, 95% CI 1.0, 13.5). (Note that in ExternE 1995, odds ratios for children without a history of asthma were mistakenly quoted, rather than the values above. However, in ExternE 1995 the above values of 2.5 and 3.7 were used in the derived calculations. This was as intended. Consequently the derived values below are unchanged from ExternE 1995.)

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. Indeed, Dockery *et al*. conclude that most of the extra episodes of bronchitis occurred among those children with a history of asthma; and so there is an element of double-counting in including this endpoint in addition to those for acute effects in asthmatic children described earlier. This however was not the case for cough.

Re-expressing the odds ratios (of 2.5 and of 3.7, respectively) in terms of unit exposure gives, for bronchitis, OR = 1.024 (95% CI = 1.003, 1.048) and, for chronic cough, OR = 1.034 (95% CI = 1.0, 1.069), for 1 $\mu g/m^3$ increase of PM_{15}. Combining 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; and scaling upwards to convert from PM_{15} to PM_{10} using the factors PM_{10} = 0.9 PM_{15} (Dockery and Pope, 1994) gives the following annual increments per $\mu g/m^3$ of PM_{10}:

- Increase in prevalence of children with bronchitis per 100,000 per year = 161 per $\mu g/m^3$ PM_{10};

- Increase in prevalence of children with chronic cough per 100,000 per year = 207 per $\mu g/m^3$ PM_{10}.

8.7 Additivity

Within endpoint, effects are additive unless otherwise stated.

Between endpoints for particles, assume that all respiratory hospital admission days (RHA), congestive heart failure admissions (CHF), and cerebrovascular admissions (CVA) are also restricted activity days (RAD). It is worth understanding a little more clearly what these effects really involve. Respiratory admissions include asthma, pneumonia, bronchitis and COPD. Congestive heart failure is a condition relating to the ability of the heart to pump blood and is more prevalent in the elderly. Cerebrovascular admissions include strokes and transient ischaemic attacks. We have looked at the number of RADs likely to be involved in each case. through an estimate of the average length of stay in hospital for each type of admission. A value of 9.5 days was used by Rowe *et al* (1994) for respiratory admissions, derived from US hospital usage data. We therefore propose a value of 10 days for the average length of stay for a respiratory condition. We propose an average stay of 7 days for a congestive heart failure admission and 45 days for a cerebrovascular admission.

- Hence net RAD = RAD - (RHA * 10) - (CHF * 7) - (CVA * 45).

Bearing in mind the number of cases of each type involved, the adjustment to RADs is negligible, but worth making to show that the work has been done thoroughly.

Between endpoints for ozone, all asthma attack days (AA) are also MRADs. The function for MRADs refers to adults (80%) only, while that of AA refers to all asthmatics (3.5%).

- Hence net MRAD = MRAD - (AA * 0.8 * 0.035).

8.8 Uncertainty

As discussed in more detail in section 3.9 uncertainty is expressed in terms of uncertainty labels for each impact, somewhat analogous to the H, M and L confidence levels used in the first series of ExternE reports (EC 1995). However, we now use a more quantitative definition based on geometric standard deviations σ_G and confidence intervals of the lognormal distribution. The labels are:

A = high confidence, roughly corresponding to σ_G = 2.5 to 4;

B = medium confidence, roughly corresponding to σ_G = 4 to 6;

C = low confidence, roughly corresponding to σ_G = 6 to 12.

In some cases a '?' is added to indicate cases where the damage cost may well be lower than our estimate.

Table 8.1 Quantification of human health impacts. The exposure response slope, f_{er}, is for Western Europe and has units of [cases/(yr-person-µg/m^3)] for morbidity, and [%change in annual mortality rate/(µg/m^3)] for mortality.

Receptor	Impact Category	Reference	Pollutant	f_{er}	Uncertainty rating
ASTHMATICS (3.5% of population)					
adults	Bronchodilator usage	Dusseldorp *et al*, 1995	PM_{10},	0.163	B
			Nitrates,	0.163	B?
			$PM_{2.5}$,	0.272	B
			Sulphates	0.272	B
	Cough	Dusseldorp *et al*, 1995	PM_{10},	0.168	A
			Nitrates,	0.168	A?
			$PM_{2.5}$,	0.280	A
			Sulphates	0.280	A
	Lower respiratory symptoms (wheeze)	Dusseldorp *et al*, 1995	PM_{10},	0.061	A
			Nitrates,	0.061	A?
			$PM_{2.5}$,	0.101	A
			Sulphates	0.101	A
children	Bronchodilator usage	Roemer *et al*, 1993	PM_{10},	0.078	B
			Nitrates,	0.078	B?
			$PM_{2.5}$,	0.129	B
			Sulphates	0.129	B
	Cough	Pope and Dockery, 1992	PM_{10},	0.133	A
			Nitrates,	0.133	A?
			$PM_{2.5}$,	0.223	A
			Sulphates	0.223	A
	Lower respiratory symptoms (wheeze)	Roemer *et al*, 1993	PM_{10},	0.103	A
			Nitrates,	0.103	A?
			$PM_{2.5}$,	0.172	A
			Sulphates	0.172	A
all	Asthma attacks (AA)	Whittemore and Korn, 1980	O_3	4.29E-3	B?
ELDERLY 65+ (14% of population)					
	Congestive heart failure	Schwartz and Morris, 1995	PM_{10},	1.85E-5	B
			Nitrates,	1.85E-5	B?
			$PM_{2.5}$,	3.09E-5	B
			Sulphates,	3.09E-5	B
			CO	5.55E-7	B

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Table 8.1 (continued). Quantification of human health impacts.

Receptor	Impact Category	Reference	Pollutant	fer	Uncertainty rating
ADULTS (80% of population)					
	Restricted activity days (RAD)[1]	Ostro, 1987	PM_{10},	0.025	B
			Nitrates,	0.025	B?
			$PM_{2.5}$,	0.042	B
			Sulphates	0.042	B
	Minor restricted activity day (MRAD)[2]	Ostro and Rothschild, 1989	O_3	9.76E-3	B
	Chronic bronchitis	Abbey *et al*, 1995	PM_{10},	4.9E-5	A
			Nitrates,	4.9E-5	A?
			$PM_{2.5}$,	7.8E-5	A
			Sulphates	7.8E-5	A
ENTIRE POPULATION					
	Respiratory hospital admissions (RHA)	Dab *et al*, 1996	PM_{10},	2.07E-6	A
			Nitrates,	2.07E-6	A?
			$PM_{2.5}$,	3.46E-6	A
			Sulphates	3.46E-6	A
		Ponce de Leon, 1996	SO_2	2.04E-6	A
			O_3	3.54E-6	A
	Cerebrovascular hospital admissions	Wordley *et al*, 1997	PM_{10},	5.04E-6	B
			Nitrates,	5.04E-6	B?
			$PM_{2.5}$,	8.42E-6	B
			Sulphates	8.42E-6	B
	Symptom days	Krupnick *et al*, 1990	O_3	0.033	A
	Acute Mortality (AM)	Spix *et al*, 1996, Verhoeff *et al*, 1996	PM_{10},	0.040%	B
			Nitrates,	0.040%	B?
			$PM_{2.5}$,	0.068%	B
			Sulphates	0.068%	B
			SO_2	0.072%	B
		Anderson *et al*, 1996, Touloumi *et al*, 1996 Sunyer *et al*, 1996	O_3	0.059%	B
	Chronic Mortality (CM)	Pope *et al*, 1995	PM_{10},	0.39%	B
			Nitrates,	0.39%	B?
			$PM_{2.5}$,	0.64%	B
			Sulphates	0.64%	B

[1] Assume that all days in hospital for respiratory admissions (RHA), congestive heart failure (CHF) and cerebrovascular conditions (CVA) are also restricted activity days (RAD). Also assume that the average stay for each is 10, 7 and 45 days respectively.

Thus, **net RAD = RAD - (RHA*10) - (CHF*7) - (CVA*45)**.

[2] Assume asthma attacks (AA) are also minor restricted activity days (MRAD), and that 3.5% of the adult population (80% of the total population) are asthmatic.

Thus, **net MRAD = MRAD - (AA*0.8*0.035)**.

Table 8.2. Human health E-R functions for *sensitivity analysis only* (Western Europe). The exposure response slope, f_{er}, is for Western Europe and has units of [cases/(yr-person-µg/m^3)] for morbidity, and [%change in annual mortality rate/(µg/m^3)] for mortality.

Receptor	Impact Category	Reference	Pollutant	fer[1]	Uncertainty rating
ELDERLY, 65+ (14% of population)					
	Ischaemic heart disease	Schwartz and Morris, 1995	PM_{10},	1.75E-5	B
			Nitrates,	1.75E-5	B?
			$PM_{2.5}$,	2.92E-5	B
			Sulphates	2.92E-5	B
			CO	4.17E-7	B
ENTIRE POPULATION					
	Respiratory hospital admissions (RHA)	Ponce de Leon, 1996	NO_2	1.40E-6	A?
	ERV for COPD	Sunyer *et al*, 1993	Nitrates, PM_{10}	7.20E-6	B?
			Sulphates, $PM_{2.5}$	1.20E-5	B?
	ERV for asthma	Schwartz, 1993 and Bates, 1990	Nitrates, PM_{10}	6.45E-6	B?
			Sulphates, $PM_{2.5}$	1.08E-5	B?
		Cody, 1992 and Bates, 1990	O_3	1.32E-5	B?
	ERV for croup in pre school children	Schwartz *et al*, 1991	Nitrates, PM_{10}	2.91E-5	B?
			Sulphates, $PM_{2.5}$	4.86E-5	B?
	Cancer risk estimates	Pilkington *et al*, 1997	Formaldehyde	1.43E-7	B?
	Acute Mortality (AM)	Touloumi *et al*, 1994	CO	0.0015%	B?
		Sunyer *et al*, 1996, Anderson *et al*, 1996	NO_2	0.034%	B?

[1] Sources: [EC, 1995] and [Hurley *et al*, 1997].

Additional suggested sensitivity analyses:
(1) Try omitting SO_2 impacts for acute mortality and respiratory hospital admissions;
(2) Treat all particles as PM_{10} or $PM_{2.5}$;
(3) Try omitting all RADs and MRADs;
(4) Scale down by a factor 2 the E-R functions for chronic mortality by Pope *et al* .

8.9 References

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9. health effects of heavy metals, dioxins, and other atmospheric organic pollutants

9.1 Introduction

A characteristic of micropollutants such as 'dioxin' and heavy metals is that they are persistent in the environment. In consequence they are available to receptors both directly through inhalation and indirectly over the longer term through contact with soil, food and water that has become contaminated by deposited material.

The basic impact assessment approach used in ExternE for macropollutants is still valid for the micropollutants - after all it simply seeks to quantify the pathway from emission to impact and monetary damage. However, the step in which incremental exposure of the stock at risk is quantified requires elaboration to account for both direct and indirect exposure. The range of possible exposure pathways is shown in Table 9.1.

Table 9.1 Exposure pathways for persistent micropollutants.

Direct Exposure	Indirect Exposure
Inhalation	Ingestion of contaminated food
	Ingestion of contaminated water
	Ingestion of contaminated soil
	Dermal contact

In consequence, total exposures are dependent much more on local conditions, behavioural factors, etc. than for the macropollutants. Reflecting this, analysis of the effects of micropollutants is typically conducted over a restricted region - that in which impacts from a given plant are thought to be most likely. The scope of ExternE, however, requires the analysis to be conducted on a broader base than this, requiring conclusions to be reached from exposures across the European Union. In view of the fact that detailed modelling of exposures to micropollutants is inappropriate at such a scale (Renner, 1995), we have instead used available data on exposure levels from published reviews.

Assessment of impacts is also different for this group of pollutants. The principal concerns again surround human health effects, but these may be both carcinogenic, as in the case of cadmium, or non-carcinogenic, as in the case of mercury and lead. The relationships between exposure to most non-carcinogens and the probability of a health effect are believed to show threshold levels below which no effect will be seen. These thresholds typically vary between individuals, depending on age, genetic factors, medical history, etc. The question of thresholds for carcinogens is largely dependent on whether their effects are genotoxic or receptor-mediated, with thresholds being more relevant to the latter.

9.2 Terminology

There are terms used throughout the text which are defined here for the purpose of clarity.

Parts per million (ppm) or parts per billion (ppb) - This expresses the concentration by volume of a gas or vapour in air. Note that the unit changes dependent on the pollutant in question, reflecting wide differences in the concentrations of different species.

Milligrams per metre cubed (mg/m^3) - This expresses the concentration by mass of a substance suspended in air.

Occupational exposure standard (OES) - This is the concentration of an airborne substance averaged over a reference period, at which on the basis of current knowledge there is no evidence that it is likely to be injurious to workers exposed to this concentration on a daily basis.

Maximum Exposure Limit (MEL) - This is the maximum concentration of an airborne substance, averaged over a specified reference period, to which workers may be exposed by inhalation under any circumstances. A typical reference period is an average work shift, 8 hour TWA (time weighted average).

Unit risk factor - These are used to represent the chance of contracting cancer from a lifetime exposure (70 years) for a 70 kg person breathing in 1 $\mu g/m^3$ of a pollutant. It is also assumed that the lifetime incidence levels could be divided by 70 to represent annual incidence levels.

9.3 Dioxins and Dibenzofurans

The dioxins are a family of 75 chlorinated tricyclic aromatic compounds, to which are often added 125 closely related compounds, the polychlorinated dibenzofurans. Several of these are highly toxic and they may also be carcinogenic. Their toxicity is illustrated by concern in spite of their emission levels being of the order of pg (10^{-12} g) per Nm^3, contrasted with levels greater than µg (10^{-6}) per Nm^3 for the other air pollutants of interest.

For our purposes, analysis can be simplified using internationally accepted toxic equivalence factors (TEFs) relating the toxicity of other dioxins to 2,3,7,8-tetrachlorodibenzodioxin (TCDD) (which is believed to be the most toxic dioxin). These factors are shown in Table 9.2. The aggregate figure of dioxin emissions, referred to as the toxic equivalence quotient (I-TEQ), is calculated by summing the products of mass of emission and TEF for each species. All aggregate dioxin emission data presented in this report are given in I-TEQ.

9.3.1 Threshold levels

There is considerable debate about thresholds for the effects of dioxins on human health. Of particular note is the apparent divergence in opinion between Europe, where thresholds for carcinogenic and non-carcinogenic impacts of dioxins are generally accepted, and the USA, where no (or extremely low) threshold is assumed. Positions on both sides of the Atlantic are under review. Recent reviews for governments in France, the UK, and Germany all concluded that a threshold exists.

Table 9.2. Toxic equivalent factors for dioxins, relative to that of TCDD 2,3,7,8 (NATO/CCMS, 1988).

	Abbreviation	**TEF**
1. Tetrachlorodibenzodioxin 2,3,7,8	TCDD	1
2. Pentachlorodibenzodioxin 1,2,3,7,8	PeCDD	0.5
3. Hexachlorodibenzodioxin 1,2,3,4,7,8	HxCDD	0.1
4. Hexachlorodibenzodioxin 1,2,3,7,8,9	HxCDD	0.1
5. Hexachlorodibenzodioxin 1,2,3,6,7,8	HxCDD	0.1
6. Heptachlorodibenzodioxin 1,2,3,4,6,7,8	HpCDD	0.01
7. Octachlorodibenzodioxin	OCDD	0.001
8. Tetrachlorodibenzofuran 2,3,7,8	TCDF	0.1
9. Pentachlorodibenzofuran 2,3,4,7,8	PeCDF	0.5
10.Pentachlorodibenzofuran 1,2,3,7,8	PeCDF	0.05
11.Hexachlorodibenzofuran 1,2,3,4,7,8	HxCDF	0.1
12.Hexachlorodibenzofuran 1,2,3,7,8,9	HxCDF	0.1
13.Hexachlorodibenzofuran 1,2,3,6,7,8	HxCDF	0.1
14.Hexachlorodibenzofuran 2,3,4,6,7,8	HxCDF	0.1
15.Heptachlorodibenzofuran 1,2,3,4,6,7,8	HpCDF	0.01
16.Heptachlorodibenzofuran 1,2,3,4,7,8,9	HpCDF	0.01
17.Octachlorodibenzofuran	OCDF	0.001

The position of the World Health Organisation (French Academy of Sciences, 1995) is that the tolerable daily intake (TDI) is 10 pg/kg_{bw}·day (10^{-12}g per kg body weight per day). The TDI represents an average lifetime dose, below which damage is considered unlikely. Calculation of the TDI involves the use of safety factors, which is illustrated in Table 9.3.

Table 9.3 Use of safety factors in setting guideline intake levels (DoE, 1989).

Effect	**NOEL[(a)] pg/kg_{bw}·day**	**Safety factor**	**Guideline level pg/kg_{bw}·day**
Immunotoxic	6000	100	60
Reprotoxic	120	100	1
Carcinogenic	10000	1000	10

[(a)] No observed effect level - derived from experimental data on sensitive animal species.

Safety factors reflect the uncertainty involved in extrapolating data between species and also the perceived severity of the effect.

To calculate a lower estimate for dioxin damages we take the TDI of 10 pg/kg_{bw}·day as threshold. This is considered applicable to carcinogenic as well as non-carcinogenic effects, because dioxins are believed to be receptor-mediated carcinogens.

In contrast the position adopted by the US Environmental Protection Agency is for an acceptable daily intake about 1000 times lower based on an upper bound risk assessment of the level that carries a lifetime cancer risk of one in a million. The assumption that there is no threshold can thus be adopted for estimation of an upper estimate for damages, though it is emphasised that most expert opinion in Europe would follow the assumption that a threshold exists (although there is dispute as to the magnitude of that threshold).

9.3.2 Pathway analysis for dioxins

Considerable debate has surrounded the calculation of human exposure to dioxins from incineration. Whilst early studies concentrated on the direct (inhalation) exposure route, more recent analyses have modelled the transfer of the contaminants from the incinerator to the exposed population via most, or all of the routes shown in Table 9.1.

HMIP (1996) assessed the health risk from dioxins emitted to air by hypothetical municipal waste incineration plants located in rural and urban areas of the UK. The principal scenarios were based upon a plant size of 250,000 tonnes/year, with a dioxin emission concentration of 1.0 ng I-TEQ/Nm^3, but the analysis was extended to plant ranging from 100,000 to 500,000 tonnes/year, with dioxin emissions from 0.1 to 10 ng I-TEQ/Nm^3. Municipal waste incinerators meeting the current EU Directive will mostly emit within this range, though some go further, and some may have been exempted so far from the legislation. The study considered in detail the transfer of dioxins from air concentrations, via the soil, vegetation and animal food products, and via inhalation, to the human population in the vicinity of the plant. The dose received was calculated, across all plant sizes and emission concentrations, for average cases and a 'Hypothetical Maximally Exposed Individual' (HMEI). The HMEI is assumed to be located at the point of maximum ground level air dioxin concentration, consuming food which has been grown or reared at this location, drinking water from a reservoir also sited at this location, and exposed to such conditions over their entire lifetime. The HMEI therefore provides an ultra-conservative estimate of the risks faced by an individual. The analysis covered background exposure and incremental exposure due to the incinerator. This allowed assessment of the relative importance of the different sources of the total dose, and, since the study used the WHO threshold value to assess health effects, an assessment of the net risk to the population from dioxin intake. The pathway for the analysis is shown diagrammatically in Figure 9.1. Table 9.4 summarises the results for the plant emitting the highest levels of dioxins considered by HMIP.

Table 9.4. Summary of mean dioxin intakes for an adult HMEI* living close to an incinerator sited in urban and rural locations.

Exposure	Urban Site pg I-TEQ $kg.bw^{-1}$ day^{-1}	Rural Site pg I-TEQ $kg.bw^{-1}$ day^{-1}
Background	0.96	0.96
Incremental	0.73	0.12
Total	1.69	1.08

* Plant scenario: 500,000 t/y^{-1}, 10 ng I-TEQ Nm^{-3} emission concentration

It can be seen from Table 9.4 that even in the worst case considered, of an urban HMEI living near the largest plant emission considered by the study, the total intake does not approach the WHO threshold level. However, recent studies have suggested that the dioxin intake of an average breast-fed baby could be as high as 110 pg/kg/day at two months, falling to 25 pg/kg/day at ten months. Results from the HMIP study are not directly comparable (being averaged over a longer period), but also suggest exposure above the WHO recommended TDI of 10 pg/kg/day. However, we adopt the position of recent reviews (DoH, 1995), that when averaged over a lifetime, the cumulative effect of increased dioxin intake during breast feeding is not significant. The HMIP study concluded that emissions of dioxins from municipal waste incinerators operating to EU legislative standards do not pose a health risk to individuals, irrespective of the location and size of the incinerator or the exposed population.

Since the highest emissions limits used in the above study correspond to or exceed emissions from any incinerator likely to be built within the EU, it follows that no greater health effect should be seen from plant that meet existing Directives, assuming the threshold assumption made here is correct. Therefore estimated dioxin related damages would be zero (accepting that the present analysis is necessarily performed at too coarse a scale to pick up any individuals who, for whatever reason, have a far higher exposure to dioxin than the rest of the population).

There are two difficulties here. It is possible that breast-fed infants could be particularly sensitive to dioxins because of their developmental status. It is also possible that the threshold assumption adopted here is wrong, and that there is either no threshold, or that any threshold that does exist is so low as not to make a difference (in other words it is below typical exposure levels). In view of the genuine scientific uncertainty that exists, in particular the different attitudes between informed opinion in Europe and the USA, we therefore consider it appropriate to also consider the magnitude of the effect under the alternative assumption that there is no threshold (this would cover the full range of outcomes). Our view is that this is unlikely, but that the possibility cannot be excluded given the peculiar nature of dioxins (being present at minute levels, but having a very high toxicity). In this case it is not appropriate to restrict the analysis to the area in the vicinity of an incinerator, or to most exposed individuals. Everyone at risk of exposure from the specified plant should be considered. In practice this means consideration is given to people exposed to minuscule incremental levels of pollution. The probability of any individual being affected is small. However, the total risk summed across the exposed population may well be significant.

For this sensitivity analysis we do not, however, consider it appropriate to carry out a full detailed assessment of all intake pathways, following the same level of detail as the HMIP study. This would be complicated by the necessary range of the assessment. Instead it is possible to simplify the analysis by calculating direct intake and multiplying this by an appropriate factor to obtain the total incremental dioxin dose. It is acknowledged that the uncertainty associated with this approach is significant. This uncertainty is reflected by the fact that the direct intake pathway provides only a small percentage of the total intake. The review by the US EPA (1994) cites a figure of 2% of the total dose arising through inhalation. Other published estimates are of a similar magnitude. This figure is assumed here to be the best available estimate. Total incremental exposure is thus calculated by multiplying inhaled dose by 50.

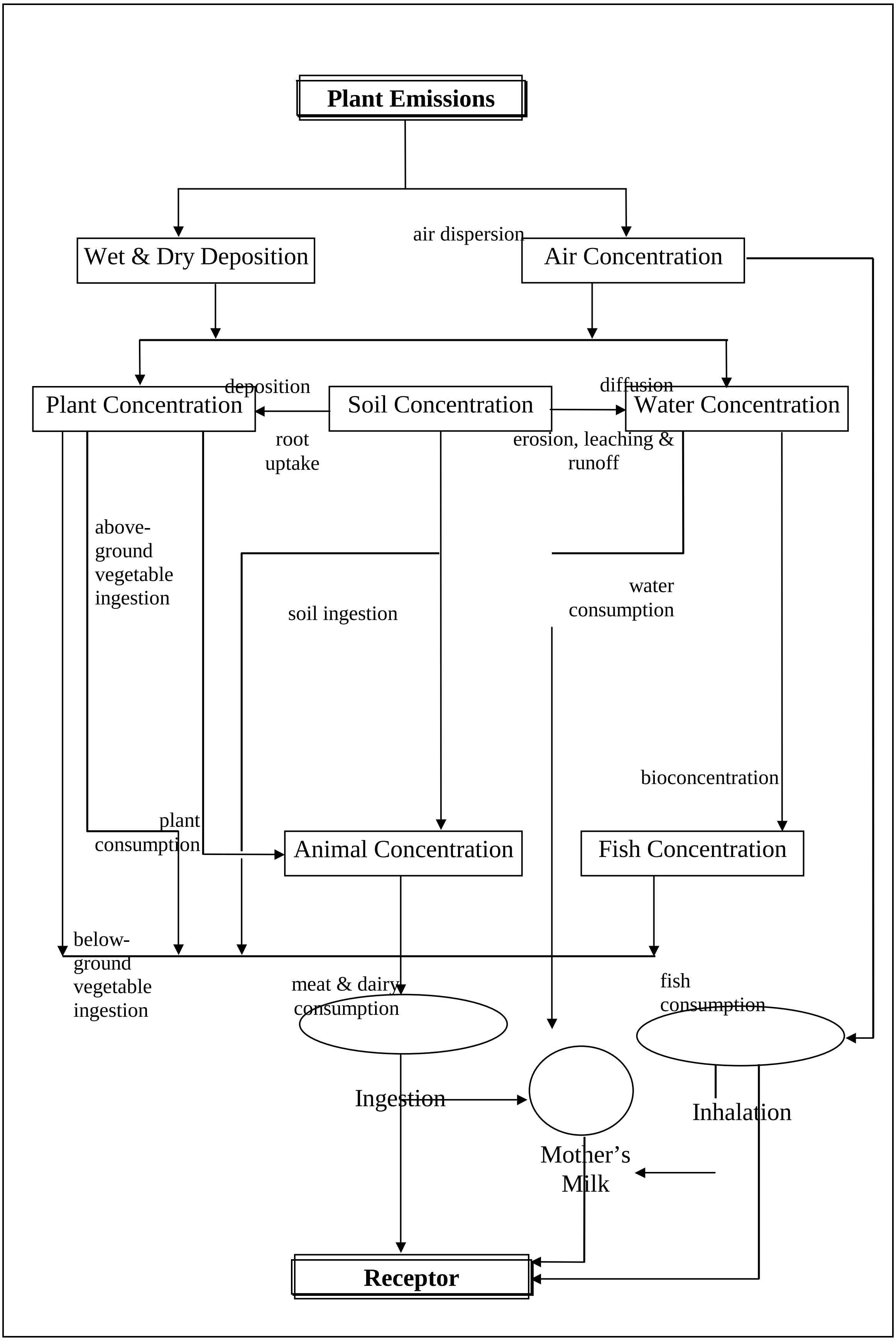

Figure 9.1 Pathway for analysis used by HMIP (1996).

The inhaled dose is calculated from the ground level concentration by the following formula:

$$I = \frac{C \times IR \times ET \times EF \times ED}{BW \times AT} \qquad 9.1$$

Where C: concentration (mg/m^3)
IR: inhalation rate (m^3/hour)
ET: exposure time (hours/day)
EF: exposure frequency (hours/year)
ED: exposure duration (years)
BW: body weight (kg)
AT: averaging time

A continuous exposure over 70 years is assumed. The factor EF * ED/AT is therefore unity, and equation (1) becomes:

$$I = \frac{C \times (IR \times ET)}{BW} \qquad 9.2$$

From this equation, and the expression of the I-TEQ per unit body weight it is apparent that body weight needs to be accounted for. Assumed values for body weight and IR*ET are shown in Table 9.5.

Table 9.5 Assumptions for calculating inhaled dose (US EPA, 1989).

	Man	**Woman**	**Child**
Body weight (kg)	70	60	20
Inhalation volume (IR*ET) (m^3/day)	23	21	15

Assuming a 46.5%, 46.5% and 7% fraction of men, women and children respectively within the total population, it is possible to calculate a gender/age-weighted 'Inhalation Factor' IF;

$$IF = \frac{23m^3}{70kg \cdot d} \cdot 0.465 + \frac{21m^3}{60kg \cdot d} \cdot 0.465 + \frac{15m^3}{20kg \cdot d \cdot} \cdot 0.07 = 0.368m^3 \ / \ (kg \cdot d) \qquad 9.3$$

The relation between dose and concentration then is

$$I = C \cdot IF = C \cdot 0.368m^3 \ / \ (kg \cdot d) \qquad 9.4$$

However, the dose described by the above equation is the inhalation dose only. To estimate the total dose, we can use the estimates on the fraction of inhalation contributing to the total dose, as given in the IEH report. Thus, the total dose is estimated to be

$$I_{Total} = C \cdot \frac{IF}{InhalationFraction} \qquad 9.5$$

with e.g. an Inhalation Fraction of 0.02 for Dioxins (relative exposure via inhalation = 2%).

No-threshold assumption:
Unit risk factor from
LAI 1.4 per µg/m^3
leading to the following ERF implemented in EcoSense:
(1) No. of additional cancers = Δ Concentration [µg/m^3] * 1.4 * Population /70

Threshold assumption:
WHO ‘tolerable daily intake’: 10 pg/(kg$_{BW}$·d)
Using an Inhalation Fraction of 0.02 (relative exposure via inhalation = 2%), the air concentration equivalent to the threshold dose is 5.4 E-7 µg/m^3.

Background:

UK (HMIP, 1996)	0.96 pg/(kg*d)	==>	5.22 E-8 µg/m^3
France (Rabl, 1996):			2.4 E-8 µg/m^3
Germany (LAI):	0.41 pg/(kg*d)	==>	2.2 E-8 µg/m^3

9.4 Impact Assessment for Heavy Metals

As is the case for dioxins, the heavy metals expelled from incinerators are persistent in the environment. In some cases direct and indirect exposure pathways would need to be considered. However, there is a constraint of the availability of exposure-response data that precludes assessment of any non-carcinogenic effect for most heavy metals.

Direct intake rates are calculated from ground level air concentration using the same approach as that adopted for dioxins (see above).

For those metals with a non-carcinogenic effect, the possibility of a health impact is assessed through comparison of total dose (background plus incremental) and the threshold value below which no effects will be seen. Due to the lack of dose-response data further quantification is not possible with 2 exceptions, for lead and mercury (though see notes below).

The specific approach applied to each of the heavy metals of most concern is described below. In most cases a selection of exposure-response functions are available, we suggest alternatives for sensitivity analysis. Assessments conducted so far have suggested that the effects of heavy metal emissions will be negligible, avoiding the need to identify any single function as the best available. Other heavy metals not listed here are regarded as less toxic and hence unlikely to produce effects larger than those for the elements listed here.

The general form of the exposure-response function is as follows for all cancer effects;
No. of additional cancers = Δ Concentration [µg/m^3] * unit risk factor * Population /70

The factor of 70 annualises lifetime risk (assuming an average longevity of 70 years).

9.4.1 Cadmium

Cancer

Unit risk factors from

ATSDR (1989)	0.0018 per µg/m^3
LAI	0.012 per µg/m^3

Non-carcinogenic effects
Threshold:
WHO-Guidelines (1987):
Rural areas: present levels of < 1-5 ng/m^3 should not be allowed to increase
Urban areas: levels of 10-20 ng/m^3 may be tolerated.

Background:
According to WHO (1987); '*Cadmium concentrations in rural areas of Europe are typically a few ng/m^3 (below 5 ng/m^3); urban values range between 5 and 50 ng/m^3, but are mostly not higher than 20 ng/m^3.*'

No dose-response function is available for non-carcinogenic effects, so quantification has not been performed. However, it is noted that exceedence of the WHO guidelines does happen, so effects cannot be ruled out.

9.4.2 Mercury

Cancer

Generally not classified as carcinogenic.
Non-carcinogenic effects
Threshold:
From US-EPA: 0.3 µg/m^3

Background:
WHO-Air Quality Guidelines 1987:

rural areas:	2-4 ng/m^3
urban areas:	10 ng/m^3

Reported thresholds are so much higher than background air exposures that effects linked to air emissions from fuel cycle activities seem unlikely in all places apart from those with high mercury levels associated with certain industrial processes (which may or may not be linked to the energy sector), or high historical contamination.

9.4.3 Arsenic

Cancer

Unit risk factors from

WHO (1987)	0.003 per µg/m^3
US-EPA (1996)	0.0002 per µg/m^3
LAI	0.004 per µg/m^3

Non-carcinogenic effects
Threshold:

US-EPA	0.3 µg/(kg$_{BW}$·d)

Using the equations derived above and an Inhalation Fraction of 0.004 (relative exposure via inhalation = 0.4 %), the air concentration equivalent to the threshold dose is 3.3 ng/m^3

Background:
WHO-Air Quality Guidelines 1987:

rural areas:	1-10 ng/m^3
urban areas:	< 1 µg/m^3
France (see Rabl, 1996):	1 - 4 ng/m^3
LAI (Germany)	
rural areas:	< 5 ng/m^3
urban areas:	< 20 ng/m^3

Thus it is possible that background levels might exceed threshold, but there are no exposure response functions available for impact quantification.

9.4.4 Chromium

Cancer
Unit risk factor from

WHO (1987)	0.04 per µg/m^3

Non-carcinogenic effects
Not analysed: acute toxic effects typically only occur at high levels that are typically only encountered occupationally.

9.4.5 Nickel

Cancer
Unit risk factors from

WHO (1987)	0.0004 per µg/m^3
US-EPA (1996)	0.004 per µg/m^3

Non-carcinogenic effects
Threshold:

ATSDR (1996)	0.02 mg/(kg$_{BW}$·d)

Using the equations derived above, and an Inhalation Fraction of 0.003 (relative exposure via inhalation = 0.3 %), the air concentration equivalent to the threshold dose is 0.16 µg/m^3 or 160 ng/m^3.

Background:
WHO (1987)

rural areas:	0.1 - 0.7 ng/m^3
urban areas:	3 - 100 ng/m^3

industrial areas: 8 - 200 ng/m^3

Again there is the possibility that some individuals will be exposed to levels above the threshold, though as before, in the absence of a dose-response function a quantification of damages is not possible.

9.4.6 Lead

General remarks

The toxicology of lead in animals, including developmental and carcinogenic potential has been covered by many reviews, however the US EPA Science Advisory Board (1989) concluded that there was limited understanding of the mechanisms of lead induced cancer in animals, and that it was inappropriate to extrapolate animal data for the purpose of quantitative risk assessment in humans. The US EPA (1991) summarised the current state of scientific knowledge for human exposure to environmental lead.

Acute effects of high exposures

For inorganic lead, blood lead levels in excess of 80 µg/dl for men and 40 µg/dl for women are categorised as `lead poisoning'. This condition now rarely occurs in the UK and is usually the result of incremental exposure over months, due to poor occupational hygiene controls. Classically the condition is associated with abdominal colic, degenerative brain disease, renal and nerve damage. These changes would also occur following exposure to excessive airborne lead levels for several hours. Organic lead poisoning is often associated with blood lead levels which are only moderately raised, but with high urinary lead levels. In this case, acute psychiatric symptoms predominate. However, both types of lead poisoning may occur in the same individual.

Possible human carcinogen

<u>Animal studies</u> Animal evidence suggests that lead exposure can result in the development of malignant tumours of the kidney or renal tract.
<u>Human studies</u> Human studies have suggested a small excess of respiratory cancer, although there is no clear trend with length or degree of exposure. In most cases these results could have been confounded by other exposures, including smoking.
Cooper *et al* (1985) found a statistically significant excess of stomach cancer (34 observed, 20.2 expected), and respiratory cancers (116 observed, 93.5 expected) in US battery workers followed up between 1947 and 1980. However, there was an unexplained downward trend in SMR with number of years of employment, and the excess in lead production workers was not significant. Selevan *et al* (1985) found a non-significant excess of respiratory cancer in lead smelters (28 observed, 25.7 expected in the highest exposure group). Excesses were also noted for kidney cancer (6 observed, 2.9 expected) and bladder cancer (6 observed, 4.2 expected).

Cancer risk estimate

There are no reliable estimates of risk of cancer as a result of exposure to ambient lead levels.

Other chronic health effects

Affect on red blood cell production

Blood lead levels as low as 10-15 µg/dl have been associated with reduced activity of enzymes involved in the production of red blood cells. Levels persistently above 60 µg/dl in occupational settings can be associated with anaemia.

Affect on brain and nervous tissue

Altered brain electrical activity has been identified at levels below 15 µg/dl, the full significance of this effect is not yet understood. Slowed conduction in peripheral nerves in children has been human carcinogen (Category 3). The lead-in-air standard (HSE) is an 8 hour time-weighted average concentration: For lead and lead compounds except for tetraethyl lead, the limit (as Pb) equals 0.15 mg/m^3 of air. For tetraethyl lead (organic), the limit (as Pb) equals 0.10 mg/m^3 of air.

Conclusion

It is apparent that a range of health effects are potentially associated with blood lead levels that are commonly found within the general population (< 10 µg/dl). These effects are particularly important in the developing foetus and young child. Due to the various sources of lead exposure, and the complex nature of lead metabolism and storage within the body, it is difficult to quantify the exact contribution of lead, from transport sources, on health effects. However, it is clear that due to the potential health effects at low blood lead levels, exposure from all sources should be reduced as far as reasonably practical.

9.4.7 Platinum and related compounds

Platinum (and palladium) is used in automobile exhaust catalytic converters to control air pollution.

Acute effects of high exposure

There is little data available on this aspect of potential health effects, although the information detailed below on skin and lung disease is applicable.

Allergic lung and skin disease

Animal studies Variable results have been shown on skin irritancy tests in animals dependent on the type of platinum salt used.

Human studies Allergic effects in humans as a result of inhalation of platinum salts were first documented in 1804. Hunter *et al* (1945) described nasal allergy in platinum refinery workers , where airborne exposures ranged from 0.9-1700 µg/m^3 . Roberts (1951) described `platinosis', a progressive allergic reaction which leads to pronounced asthmatic symptoms, in platinum refinery workers. Approximately 60% of workers exhibited asthmatic symptoms and 40% had mild dermatitis. Levene (1971) also reported allergic skin disease in platinum refinery workers. Once allergy develops in occupational settings, as platinum is a potent sensitiser, prevention of further exposure is the only solution to prevent worsening symptoms, although symptoms are likely to persist even when exposure ceases.

Current regulatory limits

Platinum is listed by the HSE as a known sensitiser, and has an OES of 5 mg/m^3 for platinum metal and 0.002 mg/m^3 for soluble platinum salts. These limits are based on available evidence, and are set to limit the sensitisation potential.

Conclusion

There is little available evidence on the airborne levels of platinum and related salts which arise due transport sources. However due to dilutional effects in the atmosphere, levels are likely to be well below current occupational standards, and health effects are unlikely to occur from this source alone.

9.5 Impact Assessment for (Mainly) Transport-Related Organic Pollutants

9.5.1 Aldehydes (including formaldehyde and acrolein)

Airborne levels of aldehydes are not routinely measured in ambient air. When measured, they are usually expressed in terms of total aldehyde concentration (of which 50% is typically formaldehyde). Concentrations, based on US populations, are typically less than 50 ppb, with levels of 10 to 30 ppb in urban areas, and as low as 0.8 ppb in rural areas. Much higher levels are found indoors or in occupational settings. Again based on US studies, the average increment to indoor formaldehyde levels as a result of passive smoking is approximately 0.25 ppm, or 250 ppb.

a. Formaldehyde

Acute effects of high exposures

<u>Animal studies</u> The major non-cancer effects in animals are due to the irritant effects of formaldehyde. Mucosal irritation in rats has been seen after only a few days exposure to 0.5 ppm.

<u>Human studies</u> Formaldehyde is highly water soluble and most of inhaled formaldehyde is deposited in the lining of the nose. Airborne concentrations shown to be irritant in humans:

> *Irritant to eyes:* 0.05-2.0 ppm - worse with continuous exposure. Most studies are based on exposures over an 8 hour working day, although studies in volunteers have reported effects with exposures of less than 6 minutes, for example Bender *et al*, 1983.
>
> *Irritant to upper respiratory tract:* 0.1-25 ppm - worse with intermittent exposure.

Bernstein *et al* (1984) in a review article, reported findings of the US Scientific Assembly on Environmental and Occupational Health (1984), where 50% of the general population would develop transient effects at exposures of 0.05-1.5 ppm for periods of 8 hours or less.

Possible human carcinogen

There is suggestive evidence of cancers at various sites.

<u>Animal studies</u> Nasal cancers have been demonstrated in rodents exposed to formaldehyde in air (Rats exposed to 14.3 ppm for 30 hours/week for 24 weeks, Kerns *et al*, 1983). The dose-response relationship is markedly convex, suggesting irritation or cell damage promoting the tumour production. Unlike humans, rats and mice breathe only through their noses, and the air flow pattern in the rat nose results in high doses being delivered to certain tissues. Risk estimates based on animal studies may therefore over-estimate the actual risk to humans. Animal studies have not shown convincing evidence of tumours at other sites, although studies have reported liver damage at exposures to 3 ppm for 6 months.

<u>Human studies</u> Epidemiological studies have mostly produced inconclusive results. A mortality study of a cohort of formaldehyde workers by Blair *et al*, which showed little evidence of increased mortality, was criticised for failure to consider the `healthy worker' effect. A subsequent follow up by Sterling *et al* (1988) found a significantly increased risk for lung cancers and for all cancers as a function of cumulative exposure (using average known exposure levels, job category, and duration of exposure). Relative risks for lung cancer were increased

above cumulative exposures greater than 1 ppm. Blood, lymphatic and prostate cancers have also been reported (Stayner, 1988). There is no convincing evidence of nasal tumours in humans as a result of formaldehyde exposure, although evidence suggests a link with throat cancer.

Cancer risk estimate

The Environmental Protection Agency (EPA) risk assessment for formaldehyde gave a unit risk factor of $1x10^{-5}$ per $\mu g/m^3$ (US EPA, 1990).

Lung or skin sensitisation: Acute effects:

Lung function in Asthmatics: Human chamber studies of asthmatics have shown a reduction in lung function (20% fall $FEV_{1.0}$) after inhaling 2 ppm for 40 minutes (Witek *et al*, 1987).
Lung function in healthy subjects*:* In some other chamber studies, healthy subjects showed little change in lung function after inhaling 3 ppm for 3 hours (Kulle *et al*, 1987), although Horvath *et al* (1988) noted small across shift changes in lung function in workers exposed to <0.5 ppm for eight hours.

Lung or skin sensitisation: Chronic (non-cancer) effects:

Skin sensitisation*:* Formaldehyde is a skin irritant and causes allergic contact dermatitis, the two conditions being difficult to distinguish clinically. There are numerous reports from a range of industrial settings (Maibach *et al*, 1983), although a threshold for the induction of allergic skin effects has not been identified. The reports of allergy mainly relate to direct skin contact, although skin irritancy with gaseous formaldehyde is plausible.
Development of lung allergy*:* It has been suggested that formaldehyde is a cause of occupational asthma at inhaled workplace concentrations, although the evidence for this remains equivocal. It has often been found that reported cases had been exposed to concentrations higher than the normal range, for example at concentrations of 0.6-1.5 ppm (Grammer *et al*, 1993). Dykewicz *et al* (1991) studied immune responses in a large population from mobile homes, buildings insulated with urea-formaldehyde foam, hospital and aircraft workers, and failed to show an immunological basis for respiratory or conjunctival symptoms after gaseous formaldehyde exposure. The lack of an allergic immune response has also been reported by other authors. Several chemicals have been implicated in causing an `asthma-like' condition (Reactive Airways Dysfunction Syndrome), the symptoms resembling asthma but without producing a specific immune response. It is possible that formaldehyde could produce RADS if further evidence fails to confirm it's potential as a lung allergen.
Irreversible effects on lung function: There are several studies considering populations exposed to either formaldehyde alone, or in combination with other substances at concentrations of <0.02 to 5 ppm for several years. In keeping with other studies, Horvath *et al* (1988) found that after 10 years exposure to 0.69 ppm, particle board workers experienced no greater incidence of chronic (irreversible) decrements in lung function than a comparable control group.

Reproductive and developmental effects

There is lack of conclusive evidence that formaldehyde exposure causes any adverse reproductive or developmental effects in animals or humans. It is very unlikely that formaldehyde would produce detrimental effects in the developing foetus.

Current regulatory limits

Formaldehyde is classified as IARC Category 2A, probable human carcinogen (IARC Monographs, Volume 62, 1995). The HSE MEL is 2 ppm 8 hour TWA.

Conclusion

Formaldehyde is a potent irritant, and no clear threshold has been defined for these effects. Indoor sources of formaldehyde are likely to be of more significance when determining health effects, although it is likely that ambient levels could produce irritant symptoms to the eyes and respiratory tract in a sub-group of the general population. Formaldehyde is unlikely to cause asthmatic symptoms in healthy subjects at exposures encountered in environmental settings, but could potentially exacerbate symptoms. There is no evidence that ambient formaldehyde levels would be carcinogenic in humans.

b. Acetaldehyde and other aldehydes

Acute effects of high exposures

Irritant effects are reported as for formaldehyde, although the threshold for onset and incremental effects is less well documented than for formaldehyde.

Possible human carcinogen

<u>Animal studies</u> have suggested tumour formation at similar sites, although the evidence is less clear cut than with formaldehyde. Most have been shown to be genotoxic (gene damage).

Cancer Risk Estimate

The Environmental Protection Agency (EPA) risk assessment for acetaldehyde gave a unit risk factor of $2x10^{-6}$ per $\mu g/m^3$ (US EPA, 1990).

Non-malignant lung disease

<u>Animal Studies:</u> Other aldehydes such as acrolein (product of diesel exhaust) which are less water soluble than formaldehyde penetrate deeper in the respiratory tract. Acrolein has been shown to cause persistent changes in airway responsiveness and structural change in lung tissue at concentrations of 0.3 ppm or more (Feron *et al*, 1978).

Current regulatory limits

Acetaldehyde and acrolein both have an occupational exposure standard (OES) of 0.1 ppm. Acetaldehyde is classified as IARC category 2B (possible human carcinogen), and acrolein as Category 3, due to inadequate evidence for carcinogenicity in humans and animals at the present time.

9.5.2 Ethene

General remarks

This substance is metabolised in both animals and humans to ethylene oxide (probable human carcinogen). However this metabolic pathway becomes saturated at tissue concentrations corresponding to 5 ppm exposure to ethylene oxide. in general human studies on the effects of ethylene oxide are confounded by other chemical exposures, or small numbers of cases of disease. Ethene (ethylene) has been used as a gaseous anaesthetic, and therefore can cause asphyxia by lowering oxygen concentration. No other occupational data specifically relates to ethene exposure, however a variety of disorders are observed when exposure is to a mixture of ethene and ethylene oxide.

9.5.3 Ethylene oxide

Acute effects of high exposures

Human Studies short term high exposures (more than 800 ppm for 4 hours) cause irritant effects to the eyes, nose, throat, and lungs, with associated headache, nausea, and loss of co-ordination.

Possible human carcinogen

Animal studies Rats exposed to 50 ppm over a 2 year period had a significantly increased incidence of a type of leukaemia. Monkeys exposed to 50 and 100 ppm did not show similar effects. Mesotheliomas were also found in rats at similar exposure levels.)
Human studies Cancer mortality was studied in 3 cohorts of Swedish workers exposed to ethylene oxide, either as continuous exposure or as intermittent. Of the 733 workers included in the study, there were 49 deaths observed where 42.2 were expected. There was a 10 fold increase in leukaemia and an 8 fold increase in stomach cancer compared with Swedish national rates (Hoegstedt, Divine *et al*, 1986).

Cancer risk estimate

The US Environmental Protection Agency risk assessment for ethylene oxide gave a unit risk factor of $1x10^{-4}$ per $\mu g/m^3$ (US EPA, 1990). It is estimated that around 5% of inhaled ethene is metabolised to ethylene oxide.

Neuromuscular toxicity

Animal studies Toxic effects have been produced in nerve and muscle tissue in some species, including monkeys, at exposures above 50 ppm. Acute effects have also been reported in lung, kidney and liver.
Human studies There is limited evidence for similar effects in humans, and in the cases reported exposure is not well quantified . However, Fukushima *et al* (1986) reported loss of sensation and loss of co-ordination in workers who were sterilising equipment, the symptoms clearing gradually when exposure stopped.
A small cross-sectional study of hospital sterile supply workers (Klees *et al*, 1990) suggest subtle effects on cognitive function and nerve conduction may occur due to chronic exposures in excess of 15 ppm. Some authorities consider that an exposure limit of 1 ppm will not protect all workers from these effects.

Possible reproductive effects

Animal studies Pregnant rats inhaling ethylene oxide showed a reduction in weight gain, but

Current regulatory limits

Ethene is classified as IARC Category 3 (not carcinogenic to humans). There is no occupational exposure standard for ethene, although its asphyxiant properties are noted. Ethylene oxide is classified as IARC category 2A (probable human carcinogen), and has a maximum exposure limit of 5 ppm.

Conclusion

It is clear that the metabolites of ethene have been linked with health effects in humans. Whilst it is unlikely that these effects would be significant at current ambient ethene levels, epidemiological data indicate that workers exposed to higher levels of ethylene oxide have an excess of leukaemia and stomach cancer.

9.5.4 MTBE (methyl tertiary-butyl ether)

General remarks

MTBE is an oxygenated fuel additive, and along with similar substances such as methanol its use is likely to increase over the next few decades. For MTBE, the health effects information is derived almost exclusively from studies in rats, and exposure data is limited to a few measurements from petrol filling stations, where refuelling produces exposures of 1 ppm. Based on the available data, it appears unlikely that the general population is at high risk of exposure to MTBE, although further work is required in this area.

Acute effects of high exposures

Animal experiments in rats and non-human primates suggest disturbance of balance and sedative effects at exposures of 4000 ppm for 6 hours. The effects appear to resolve when exposure ceases.

Possible liver and kidney damage

In animal experiments these effects are only produced at exposures in excess of 1000 ppm for several weeks, which would not be expected to occur from general population exposure to ambient levels.

Possible reproductive or foetal developmental effects

In animal studies effects on maternal weight gain are only seen at exposures in excess of 4000 ppm for the duration of pregnancy, and abnormalities in the developing foetus are seen at similar concentrations. Again it would be difficult to envisage similar exposures occurring in the general population.

Possible toxic effects to brain and nervous system

Altered states of arousal in rats have been seen at exposures between 800 to 4000 ppm for 6 hours per day for 13 weeks. There have been no reports concerning the neurotoxic effects of MTBE in humans due to occupational exposure.

Current regulatory limits

At the present time there is no occupational exposure limit for MTBE, however this is under review by the HSE Advisory Bodies (ACTS/WATCH).

Conclusion

The available data from rodents suggests MTBE is of relatively low toxicity. At current occupational exposures, it is unlikely that MTBE would be a significant hazard to healthy individuals under normal conditions of use. However the likely increased usage of such substances should be considered when considering potential health effects in the general population.

9.5.5 Volatile organic compounds (VOCs)

General remarks

This is the name given to a wide range of volatile hydrocarbons, for example alkylbenzene, toluene, alkenes, formaldehyde and trichloroethane. They are categorised by the World Health Organization into 4 categories on the basis of their boiling point range, with no sharp cut off between the categories. In addition to industrial and transport sources, VOCs are emitted by buildings and furnishings, human metabolism, and almost any human activity. In non-industrial environments, each component rarely exceeds concentrations of 50 $\mu g/m^3$ (100-1000 times lower than the relevant OESs). The total concentration of all VOCs is usually <1 mg/m^3. Due to the contribution from indoor sources, many VOCs have been shown to occur in indoor environments in higher concentrations than ambient air.

Acute exposure effects

At present there is no standardised way to summarise the combined effects of the many different compounds in the atmosphere. A tentative dose-response relationship has been suggested by Mølhave (1991) based on information from studies of indoor environments, however nasal and eye irritation have been reported during episodes of photochemical pollution, and it is possible that certain VOCs contribute.

Possible human carcinogen

Some VOCs are categorised as probable human carcinogens by IARC. There is no evidence about the carcinogenic potential of this mixed group of substances, and it is difficult to predict possible health impacts as these would vary dependent on the percentage contribution of each of the substances.

Cancer risk estimate

Risk estimates have not been calculated for VOCs as a group, although there are several existing estimates for specific VOCs, for example benzene. Again the percentage contribution of each component would need to be known to enable an accurate assessment of risk, and at the present time there is inadequate data on exposure, or lack of reliable monitoring methods to provide an accurate estimate.

Current regulatory limits

In general there is a paucity of available research on VOCs from human or animal studies. There is no current OES for the group of substances as a whole although, several of the component parts (toluene OES 50 ppm 8hour TWA) do have exposure limits.

Conclusion

It is unlikely that the acute health effects reported to be associated with VOCs would occur due to sole exposure to ambient levels of VOCs within the normal range, but their presence could be additive to the effects of other irritants. There is insufficient data to provide a reliable estimate of the risk of cancer from exposure to a mixed airborne level of VOCs. It is important to disaggregate.

9.5.6 Benzene

General remarks

There is no useful data on the effects of low exposure to benzene in the concentrations normally found in ambient air. Occupational groups have been followed up since the introduction of more stringent occupational hygiene standards, although views on the latency of benzene induced leukaemia vary from 2 to 50 years, although 11 years seems a reasonable estimate based on available evidence. Automotive sources remain a substantial source of community exposure, although catalytic converters are reducing emissions. However, the removal of lead from petrol, increases the alkyl benzene component and may therefore lead to an increase in community benzene exposure. Preliminary information suggests that inhalation whilst inside a vehicle can be a significant source of total body burden, although more studies are needed.

Acute effects of high exposures

As an acute poison benzene produces narcotic effects. Human inhalation of 20,000 ppm (2% in air) is fatal in less than 10 minutes.

Human carcinogen (causes leukaemia)

<u>Animal studies</u> Bone marrow depression of red blood cell production has been seen in mice treated for 5 days at inhalational concentrations of 103 ppm. Longer term exposure in mice has produced tumours of similar types to those found in humans but only at concentrations greater than 100 ppm. Other animal species also show a variety of malignant tumours in relation to blood cell lines, and certain species have shown lung and liver cancer following benzene exposure, although the evidence for non-haematological cancers in humans is less convincing. Benzene has been shown to be clearly genotoxic (able to damage DNA) in *vivo* and *in vitro* experiments.
<u>Human studies</u> Most human studies are confounded by lack of available quantitative data, short follow up at low exposure concentrations, co-exposures to other potential carcinogens, and the fact that the body breaks down benzene to metabolites which seem to be more toxic than the parent substance. Due to this individual variation in susceptibility or metabolism may influence the risk at any given exposure.

Rinsky *et al* (1987) provided an authoritative examination of the known odds of death from benzene induced leukaemia. For an individual inhaling 1 ppm for 40 years, the odds of benzene induced leukaemic death were 1.7 times that of the unexposed worker. Therefore, the odds of benzene-induced leukaemic death at 0.1 ppm approach very nearly the odds of leukaemic death for a worker who is not exposed to benzene.
Yin *et al* (1987) in a retrospective cohort study of 28,460 workers exposed to on average 15 - 150 ppm benzene, and 28,257 non-exposed controls, found 30 cases of leukaemia in the

exposed population compared to 4 cases in controls, producing SMR of 5.74. A further study by the same author of 508,818 workers with average benzene exposures of 5.6 ppm (upper limit 308 ppm) found a 5.8 fold increase in aplastic anaemia which is classically associated with benzene exposure compared with the general population. Other studies have suggested an association with another type of haematological malignancy, multiple myeloma, although the evidence is less convincing.

In a more recent cross-sectional study (Collins *et al*, 1991) found no difference in blood results of 200 workers exposed to benzene levels in the range 0.01-1.4 ppm compared to non-exposed, over a 10 year period.

Van Damme *et al* (1991) followed up 484 employees of a petrochemical plant between 1967 and 1988. Each employee had at least 5 blood tests during this period and none had been exposed to benzene levels higher than 2 ppm over 8 hour shifts since 1971. The percentage of white blood cell counts lower than $4x10^9$/l (borderline of abnormality) was 4.3% in the study compared to 1% in a control group of non-exposed chemical plant workers. This study suggests that a sub-group of the population may be more susceptible to the effect of low benzene exposure than others. However, these findings do not necessarily suggest that such low exposures would be carcinogenic, but the outcome is different than other recent low-exposure studies. Smoking habits are one of several factors which can produce a fall in white cell count within this range.

Cancer risk estimate

The Environmental Protection Agency (EPA) risk assessment for benzene gave a unit risk factor of $8x10^{-6}$ per $\mu g/m^3$ (US EPA, 1990).

Other chronic health effects

Possible effects on brain and nervous system

Neurotoxic effects due to current occupational exposures to benzene under normal working conditions have not been reported. Animal studies show narcotic effects at exposures over 4000 ppm, subtle neuro-behavioural effects at exposures in excess of 300 ppm. In general neurotoxic effects in animals occur at exposures well above those known to produce changes within blood cell lines.

Possible hearing loss

Occupational exposure to certain solvents including benzene has been linked in several studies with the development of hearing impairment (Morata *et al*, 1993). In general these studies are difficult to interpret, as the exposures are often to a mixture of solvents, and this effect is only seen at exposures in excess of current occupational exposure limits. There also appears to be an interaction between noise and solvent exposure in causing nerve damage which results in hearing loss. It is unlikely that ambient benzene levels would be a contributory factor for hearing impairment in the general population.

Possible reproductive effects

Some studies in rats have suggested delayed foetal skeletal development, and reduction in foetal and maternal weight gain at exposures between 100-220 ppm for two weeks. There is no supporting evidence for similar effects in humans at current occupational exposure levels, and therefore these effects would be very unlikely at ambient levels generally encountered.

Current regulatory limits

Benzene is classified as Category 1 by IARC, a known human carcinogen. EPAQS (UK Expert Panel on Air Quality Standards) recommended running annual average of 5 ppb, and in view of the genotoxic nature of benzene recommended a target level of 1 ppb. For occupational exposures, the HSE has set a maximum exposure limit for benzene of 5 ppm over an 8 hour shift.

Conclusion

Whilst there is no doubt that benzene is a human carcinogen, there is little evidence to suggest an increased risk of leukaemia at the current regulatory limits for occupational exposures, or the currently lower environmental levels experienced by the general population. There is no evidence to suggest that other health effects associated with benzene exposure are likely to occur at current ambient levels.

9.5.7 1,3 butadiene

Acute effects of high exposures

<u>Human Studies</u> Early studies exposed humans to 800 ppm for 8 hours with only slight irritation to eyes and upper respiratory tract. [ILO Encyclopaedia of Occupational Health and Safety (1983) p 347-8]. However large acute exposures, butadiene acting as an asphyxiant, would lead to respiratory paralysis. Similar irritant effects have been found in animal studies, but no other acute effects.

Possible human carcinogen

1,3 butadiene is potentially carcinogenic to both the white and red cell systems. Potential for carcinogenesis should be considered separately for these two systems.
<u>Animal studies</u> 1,3-butadiene has been shown to be carcinogenic in two animal species. Tumours in rodents include haemopoietic (red cell), lymphatic (white cell) and also lung, ovarian and liver neoplasms. There is however wide discrepancy in metabolism between different species. Notably, mice metabolise butadiene monoepoxide to diepoxide which causes DNA cross-links, whereas human metabolism proceeds to a non-DNA reactive product. The level of this reactive metabolite may be correlated with the species difference (possible biomarker). Inhalation of concentrations less than 6.25 ppm causes tumours in mice (mostly lung cancer and lymphoma). Much higher concentrations (e.g. > 8000 ppm) are required for tumour production in rats.[NTP technical report series 28: NIH pubn.(1984). 84-2544].
<u>Human studies</u> 1,3-butadiene is a major ingredient of synthetic rubber, and being volatile, the route of absorption is primarily inhalation. A review of epidemiological data (Landrigan, 1990) confirmed earlier epidemiological evidence and showed the following:

1. A study of 2568 <u>workers</u> at a butadiene plant in Texas, using a qualitative exposure scale based on job codes, reported low overall mortality (SMR 84), but found excess deaths for certain rare types of lymphatic cancers (SMR 229). Excess mortality was concentrated in

workers employed less than 10 years, and in those employed during the war. Production workers were at highest risk (Downs *et al*, 1987).

2. A retrospective study of 12,113 workers employed in 8 synthetic rubber plants in US and Canada, showed mortality for all neoplasms was low (SMR 78),found excess mortality for lymphatic (white cell) and haemopoietic cancer (red cell) in production workers (Based on small numbers, SMR 507 for lymphatic cancers in black production workers). Exposure data was again qualitative based on job category (Matanoski *et al*, 1990).

Cancer Risk Estimate

The Environmental Protection Agency (EPA) risk assessment for 1,3-butadiene, gave a unit risk factor of $3x10^{-4}$ per $\mu g/m^3$ (US EPA, 1990). Based on extrapolation from animal data, and using a linear risk model, NIOSH estimated an excess incidence of cancer 597/10,000 workers with 45 years of exposure to 2 ppm. [Methodology was criticised, Science (1992) 256; 1609).

Possible risks to developing foetus

Fetotoxic effects (e.g. delayed foetal development) have been shown in rats exposed to 200 ppm in the early stages of pregnancy, but there is no evidence of teratogenesis (congenital abnormalities). No such effects have been reported in human studies, and this effect would be unlikely to arise due to ambient air concentrations of 1,3-butadiene.

Current regulatory limits: UK

It is reasonable to conclude that there is at least limited evidence for human carcinogenicity of 1,3-butadiene. 1,3-butadiene is classified by the International Agency for Research on Cancer (IARC) as Category 2a - Probable human carcinogen (IARC Monographs Volume 54, 1992). 1,3-butadiene has a maximum exposure limit (MEL) of 10 ppm 8hr TWA, but this is under review by UK Health and Safety Executive (HSE).

Conclusion

Of the health risks considered, there is evidence to suggest that 1,3-butadiene is a possible human carcinogen, and that this effect may be relevant at ambient concentrations experienced, either solely or in combination with other known human carcinogens. We know of no other health effects which might need to be considered.

9.5.8 Polycyclic aromatic hydrocarbons (PAHs)

General remarks

These are ring compounds which jointly share carbon atoms, and include a wide range of substances including benzo[a]pyrene. They are just one of the complex mixture of substances found in diesel exhaust, and result from the incomplete combustion of organic material. The relationship between benzo[a]pyrene and other PAHs is known as the `PAH profile'. This profile differs for various types of emission, but has been shown to be relatively similar in the ambient air of several towns and cities. Levels of PAHs have fallen dramatically since the 1960s, and most urban areas in Europe have levels of benzo[a]pyrene of 1-10ng/m^3. Several studies have demonstrated that the carcinogenic potency of benzo[a]pyrene (and other PAHs) is enhanced by adsorption onto particles, this may be due to the resulting increased retention time in the lungs and slowed metabolism. As PAHs including benzo[a]pyrene can be absorbed onto diesel particulates, some of the information quoted in the next section is also relevant when considering the possible health impacts of PAHs.

Acute effects of high exposures

Specific data on acute effects of high exposures to PAHs is not available, however as they may be absorbed onto the surface of other particles, they may contribute to the exacerbation of symptoms in those sub-groups of the population with pre-existing lung disease (see next section for further details).

Lung cancer

Animal studies Tokiwa *et al* (1994) investigated the mutagenic activity of PAHs, and found that tissue extracts from lung cancer patients, in which different types of PAHs could be detected were mutagenic whereas extracts from patients without lung cancer were not.
Human studies Armstrong *et al* (1994) performed a case-referent study of aluminium production workers exposed to PAHs. In the cohort of 16,297 men employed for at least 1 year, there were 338 lung cancer deaths, which represented a significant excess. Lung cancer risk increased with increasing pack-years of smoking. The maximum risk associated with benzo[a]pyrene was 2.06 (RR), at exposures of 100-299 $\mu g/m^3$ years. Higher benzo[a]pyrene exposures did not increase the risk further.

Coke oven workers have shown an increased incidence of lung and urothelial cancers.

Cancer risk estimate

Again because of the complex nature of the exposure, risk estimates are usually expressed in terms of exposure to one component, benzo[a]pyrene. The EPA unit risk factor of lung cancer for benzo[a]pyrene is $1x10^{-1}$ per $\mu g/m^3$ (US EPA, 1990). There are arguments against the use of benzo[a]pyrene as an indicator of PAH toxicity in air pollution, as some PAH is bound to particulates, and some of the gaseous components are not included by this method of analysis. Diesel exhaust emissions in particular contain a much wider range of carcinogenic components than those represented PAHs.

Current regulatory limits

In 1986 IARC and the National Cancer Institute concluded that PAHs were a risk factor for lung cancer in humans. Benzo[a]pyrene specifically rather than PAHs as a group are labelled as a probable human carcinogen, and are designated a `Risk Phrase' by the HSE (R45 - may cause cancer).

Conclusion

There is strong evidence to suggest that certain components of PAHs, for example benzo[a]pyrene are carcinogenic in humans, and that nitro-aromatics as a group pose a hazard to health. As these compounds from complex mixtures and are also absorbed onto particulates, it is difficult to quantify levels of human exposure. Ambient concentrations may indeed pose an additional cancer risk to the general population, although there is insufficient human data at the present time to provide a reliable quantification of risks.

9.5.9 Diesel exhaust particulates

General remarks

Diesel engines emit 10 times more particulates than gas engines without CATs and 100 times more than those with catalytic converters. The carbon core serves as a nucleus for the condensation of organic compounds, for example PAHs, which are by-products of combustion (totalling 10-15% of the particle by mass). 20-30% of particles with an aerodynamic diameter less than 0.25μm will be deposited in the lungs. Diesel soot has been shown to be a direct acting mutagen, that is able to cause changes in genetic material without being activated by the body's metabolism. At present the proportional contributions of the different compounds in diesel soot to total mutagenic and carcinogenic activity is not known. Initially benzo[a]pyrene was thought to be responsible for this effect, although nitro-pyrenes present in low concentrations are now thought to be largely responsible. The available US data including EPA estimations (Alder and Carey, 1989) suggest widespread environmental exposure to diesel exhaust at average soot concentrations of 0.002 mg/m^3 . The average for cities and heavily contaminated environmental sites is 0.004 mg/m^3. However in industries with heavy diesel usage such as railroad and trucking, individuals may be repeatedly exposed to diesel exhaust soot concentrations of 0.1 to 1 mg/m^3.

Acute effects of high exposures

These have only been recorded in occupational settings and at much higher levels, as indicated above, than commonly found in ambient air. These include irritant effects to eyes, and the respiratory tract and non-specific effects such as light-headedness and headache.

Possible human carcinogen

a. Lung cancer

Animal studies The IARC classification of probable human carcinogenicity is based on animal evidence. Rats develop an increased incidence of lung tumours when exposed chronically to high concentrations of diesel exhaust that overloads particle clearance defences and causes soot to accumulate, for example 25 mg/m^3 for 40 hours a week (Mauderley *et al*, 1986). However, tumours are not induced by the same exposures if the exhaust is filtered to remove the particles, and the animals are only exposed to gas and vapour phases. Results of similar studies in mice are questionable, and diesel soot is not carcinogenic in Syrian hamsters.

Human studies The weakness of the estimates of exposure is the key difficulty in estimating the health risks from the results of the numerous studies which are available. In general the evidence supports a weak but positive effect. Garshick *et al* (1988) found a small but significantly elevated risk for lung cancer in a cohort of 55,407 railroad workers. They used contemporary measures of soot associated with different jobs to classify past exposures, and incorporated some adjustment for smoking. A relative risk of 1.45 (95%CI 1.11-1.89) for lung cancer was obtained in the group with the longest duration of diesel exposure. The only other study which considered smoking as an important confounder was Boffetta *et al* (1988) who performed a hospital based case-control study. Exposure was categorised as low, possible and probable based on occupation. The odds ratio adjusted for smoking and other confounders was 0.95 (95% CI 0.78-1.16), the results therefore did not support an association between diesel exhaust and an elevated risk of lung cancer. Most other studies have assessed exposure on the basis of questionnaire to the subject or next of kin.

b. Bladder cancer

The evidence from human studies suggests a risk of the same order of magnitude as the risk of lung cancer (although the evidence from animal studies is not confirmative). Many are reported in the context of broader cancer surveys, and in many cases the confounding by cigarette smoking is not adequately considered, and often the number of cases considered is small. Three studies have been identified which include more than 50 cases and control for smoking. These include the study by Risch *et al* (1988), a case-control study considering several industries. After adjustment for smoking the odds ratio for those in contact with traffic fumes was 1.69 (p=0.008). Jensen *et al* (1987) in their case-referent study after adjustment for confounders found a relative risk of 1.3 (95% CI 1.04-1.45) in transport related industry with at least 10 years in that employment.

Cancer risk estimate

The US EPA (1994) unit risk estimate derived using a linearised multistage model is 3.4×10^{-5} based on a lifetime exposure to 1 $\mu g/m^3$ diesel particulate matter. This estimate was derived from 3 separate animal bioassays on Fischer rats.

Increase in respiratory illness

Although changes in lung function have often not reached statistical significance in studies of occupational groups. Diesel particulates are a component of the general effect of particulate air pollution on public health (mortality and morbidity) which we have reviewed elsewhere.

Other health effects

There is little evidence for other non-pulmonary health effects.

Current regulatory limits

Diesel emissions are currently classified by IARC as Category IIa probably carcinogenic to humans.

Conclusion

There is strong evidence to suggest that certain components of diesel particulates, for example PAHs are carcinogenic in humans. Diesel particulates have a complex structure, and it is difficult to quantify levels of human exposure to the component parts. Ambient concentrations may indeed pose an additional cancer risk to the general population, although there is insufficient human data at the present time to provide a reliable quantification of risks.

Table 9.5 summarises some risk factors for various organic pollutants associated primarily with transport. There is much uncertainty in several of these risk estimates, as they were derived from information on occupational exposures, requiring extrapolation from relatively short duration high exposure to lifetime duration low level exposure.

Table 9.5: Quantification of human health impacts associated with transport related organic pollutants. The risk factors shown represent change in risk per ug/m^{-3}.

Receptor	Impact Category	Pollutant	Risk factor
Entire population	Cancer risk estimates	Benzene	1.14E-7
		Benzo[a]Pyrene	1.43E-3
		1,3 butadiene	4.29E-6
		Diesel particles	4.86E-7
		Formaldehyde	1.43E-7

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10. RADIOLOGICAL HEALTH IMPACTS

10.1 Introduction

This Chapter is largely taken from Chapter 5 of the earlier ExternE Methodology report (European Commission, 1995a). As a result it is largely referenced to work that was done at that time on the French nuclear cycle by Mona Dreicer and her colleagues at CEPN.

Within this report readers should also refer to Chapter 7 on major accidents and to Chapter 12 on the valuation of health impacts. It is of course health effects of radionuclide emissions that dominate the nuclear fuel cycle; although impacts will also affect other aspects of the environment, such effects are not generally regarded to be as serious as those to health.

It is frequently said that there is inherent bias against the nuclear fuel chain in comparative risk assessments, because analysis of the nuclear fuel chain is more comprehensive than that of other fuel cycles for power generation. This no longer seems justified; detailed analysis of all power generation fuel cycles is possible, as the present report and others produced within the ExternE Project, show.

10.1.1 Boundaries of the Assessment

The most important variables for the assessment of radiological impacts concern the definition of time and space boundaries. Due to the long half-life of some 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 made within the ExternE Project 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 will remain the same as today.

The validity of this type of modelling has been widely discussed. The uncertainty of the models increases as 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 considered within the ExternE Project for which mainly shorter term impacts are considered. [Note of course that some non-radiological impacts such as those linked to greenhouse gas emissions also have extended lifetimes.] 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.

We define different time scales as follows. 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, though the most significant part of the impacts are included.

The assessment of impacts over long 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, and 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.

10.1.2 Priority Pathways for Environmental Releases

For 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 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 any 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 10.1 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.

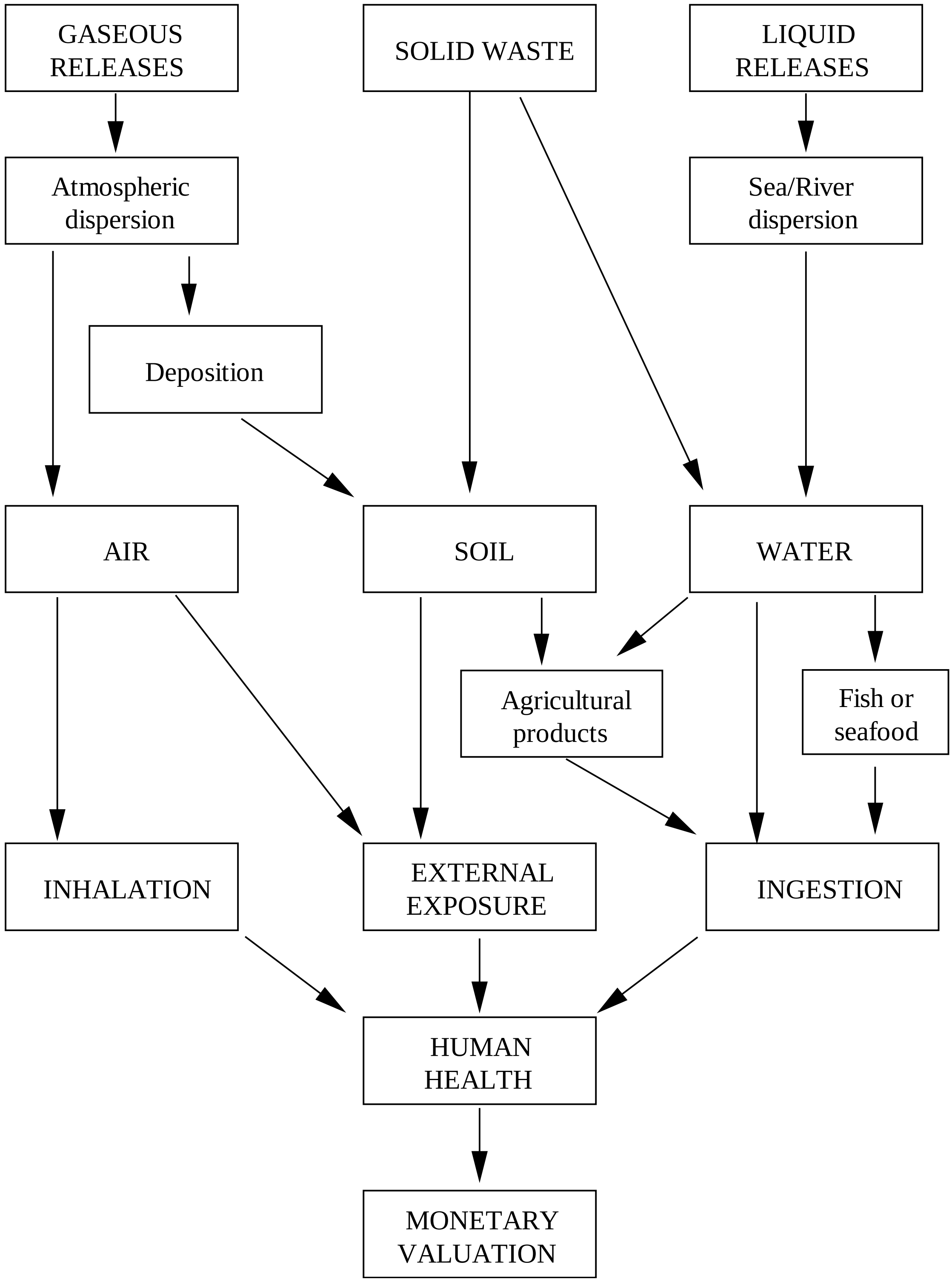

Figure 10.1 Impact Pathways for the Release of Radioactivity in the Environment

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.

10.2 Uncertainty

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.

10.3 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.

10.3.1 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).

10.3.2 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 that 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.

10.3.3 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, 1995b).

10.3.4 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 10.2 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 dependent 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 10.2 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, 1995b).

10.4 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.

10.4.1 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.

10.4.2 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.

10.5 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 sub-surface 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.

10.6 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:

$$\mathbf{Risk} = \Sigma\, \mathbf{P_i} \,.\, \mathbf{C_i}$$

10.6.1 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 10.3. The single cost value presented takes accounts for transportation between all stages of the fuel cycle. Costs of clean-up and countermeasures that may have to be taken are excluded.

10.6.2 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.

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, 1995b).

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 10.4. 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.

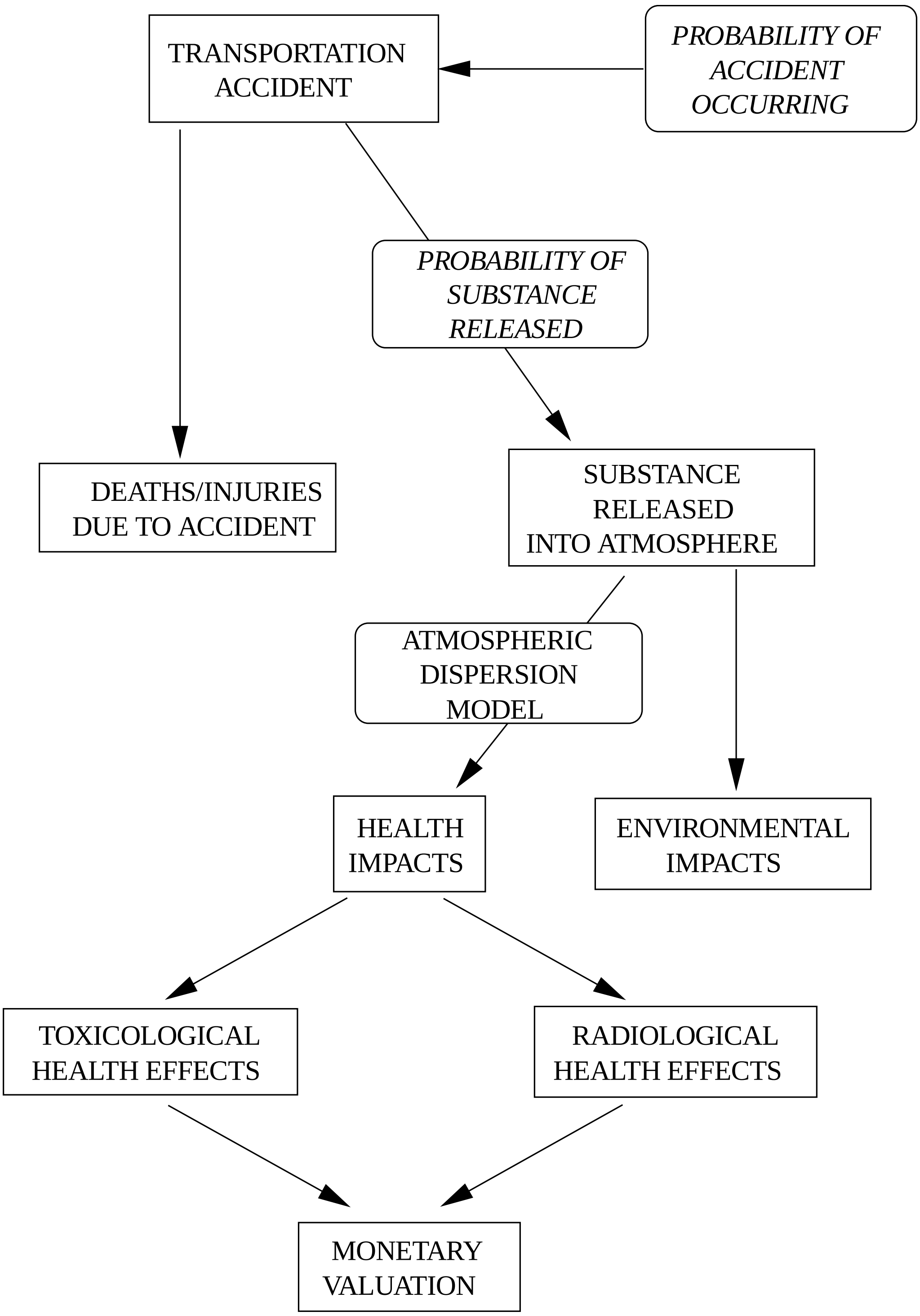

Figure 10.3 Pathway for the Assessment of a Transportation Accident.

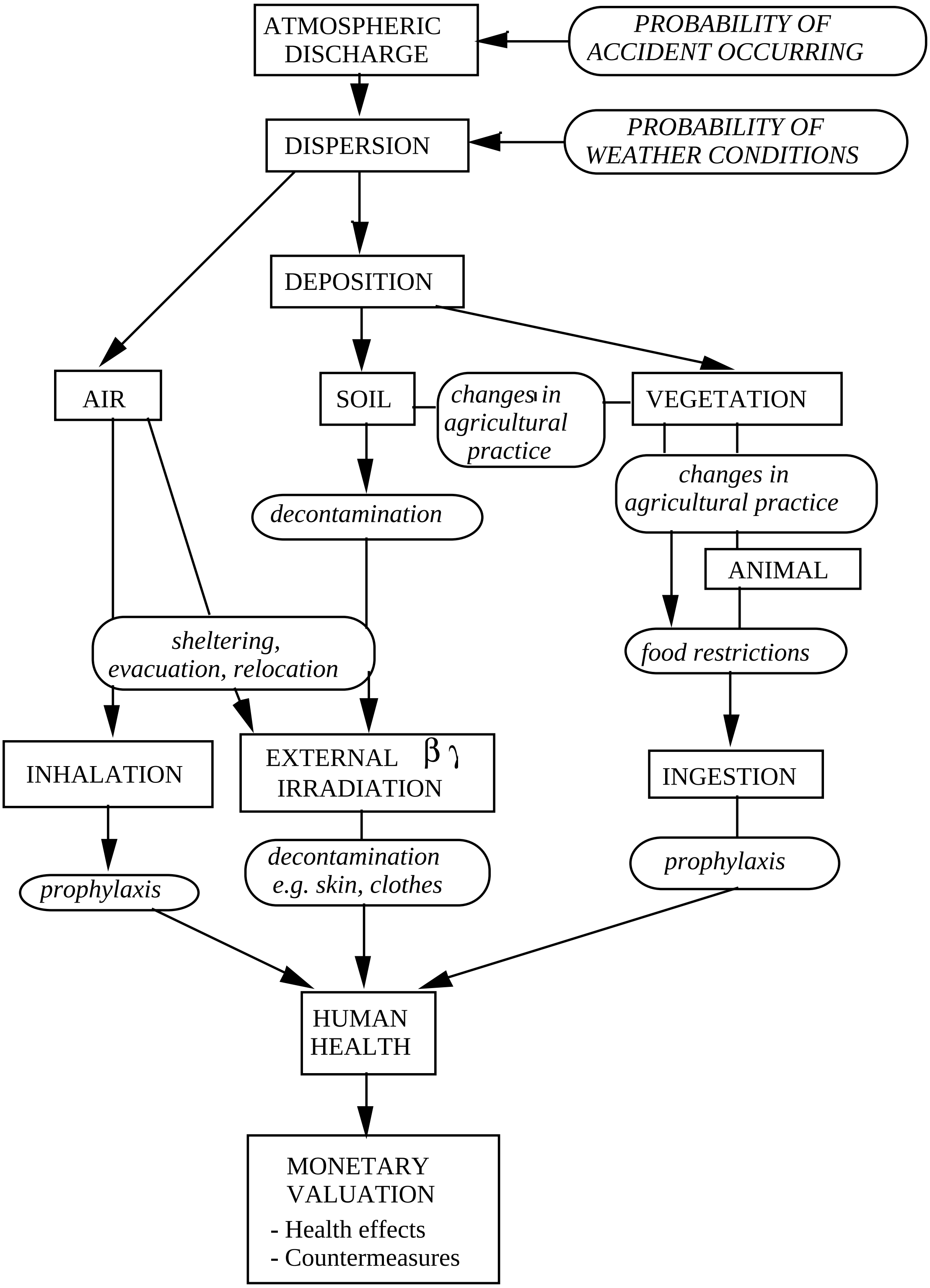

Figure 10.4 Pathways for a Severe Accidental Release.

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, 1995b).

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. This issue is described in more detail in Chapter 7.

10.7 Occupational Impacts

The assessment of the radiological and non-radiological occupational impacts is very straightforward. 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 an 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 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 accident 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).

10.8 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).

10.9 Monetary Valuation

It is assumed that the severity of a hereditary effect should be valued as the same as the Value of Statistical Life (VOSL, taken here to be 3.2 MECU, 1995 prices). 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 VOSL-severe hereditary effect values for the 3% and 10% discount rates are 0.37 MECU and 0.05 MECU, respectively.

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11. OCCUPATIONAL HEALTH EFFECTS

11.1 Introduction

Occupational health effects arise through accidents and occupational diseases. Occupational accidents are clearly both direct and instantaneous. 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 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.

11.2 Occupational Health in the Context of Fuel Cycle Analysis

Based on previous work in the Project the following phases of the fossil fuel cycles were identified as areas where occupational health effects may be significant. The list is provided simply to show how broad the analysis of these effects may have to be for any given system.

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.

For most renewable technologies, the greatest occupational risks are from the construction phases. Results of the earlier report on the nuclear fuel cycle (European Commission, 1995b) suggested that the greatest risk of exposure to radiation arises at the mining/milling and power generation phases (assuming normal operation).

Accidents during transportation may clearly affect the public as well as workers, and these should clearly be included in the assessment.

11.3 Quantifying Impacts

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.

The importance of getting the time period correct can be illustrated with reference to the Piper Alpha disaster in the UK sector of the North Sea (Taylor, 1991). The subsequent inquiry 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. Inevitably in determining the ‘correct’ period for data collection there is compromise between time period and the extent to which older data may be considered representative of current practice.

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 11.1.

Table 11.1 Reference data requirements for the estimation of occupational health impacts, using the example of the coal fuel cycle.

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

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. A global review of coal mining health data (Holland *et al*, 1998) demonstrated that data could be collected for most relevant countries. This review also demonstrated at least a factor 300 difference between the best and worst records for accidental deaths by country.

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, though available evidence suggests that health impacts are not significantly internalised in many cases.

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 11.2 provides details of the information needed for a detailed assessment of such cases.

Table 11.2 Reference data and model requirements for the estimation of health impacts in coal miners from long term exposure.

Activity	Reference data required	Type of models or relationships	Output units
Underground coal mining	Total concentration of relevant pollutant	Measured values	[mg/m^3]
Exposure	Exposure time	Sum of exposure concentrations over time	[gh/m^3]
Impact assessment	Population at risk	Dose-response functions	Cases/year

However, with the exception of coal mining (see European Commission, 1995a), a detailed damage-function approach to occupational disease has not so far been possible in the fossil and renewable fuel cycles. To a large degree, this is due to the presence of more historical data and the well characterised and severe health endpoints associated with coal miners. 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 that warrant development of a more impact-pathway based approach. In all cases, work is needed 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.

11.4 Data

Mortality and morbidity due to accidents are impacts that are best analysed from statistical data, preferably aggregated over several facilities (e.g. all within a country or region). 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 are used.

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. Five years seems to be an appropriate time span.

For illustration we focus on the UK and Germany to illustrate reporting protocols for health risks in the workplace. 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).

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 coal miners 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 in Germany are classified into:

- Number of deaths caused by occupational accidents;
- Number of deaths caused by occupational diseases;

11.6 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 coal mining (European Commission, 1995a). 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.

Over time the ExternE Project team has established a large number of different systems in different countries. Reference to the numerous reports produced under the study thus provides access to a wealth of statistical data in this area.

11.7 References

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12. HEALTH

12.1 Introduction

The literature on health impacts, both technical and economic, is strongly dependent upon studies carried out in the US. Whilst there are a number of studies on the value of mortality impacts in Europe, there have been few attempts to value impacts on morbidity. Hence reliance on the US work is inevitable, but it is important to look at the 'cultural dependence' of the estimates and methodologies more closely.

The Chapter starts with a review of the valuation of mortality impacts. This seeks to account for differences that seem likely (based on discussions between members of the study team) to occur when dealing with mortality amongst an 'average' member of the population (middle aged and in good health), compared to someone in poor health with only a limited life expectancy available. This discussion is extremely important because it affects the valuation of mortality related to air pollution, and this was shown in the previous phase of ExternE to be one of the most important sources of fuel cycle externalities. The Chapter then continues with details of the valuation of morbidity effects, and concludes by providing values for accidental deaths and injuries.

12.2 Valuing Mortality Impacts

The conventional approach for valuing mortality is based on the estimation of the willingness to pay for a change in the risk of death. This is converted into the 'value of a statistical life' (VOSL) by dividing the WTP by the change in risk. So, for example, if the estimated WTP is ECU 100 for a reduction in the risk of death of 1/10,000, the value of a statistical life is estimated at 100 * 10000, which equals one million ECU. This way of conceptualizing the willingness to pay for a change in the risk of death has many assumptions, primary among them being the 'linearity' between risk and payment. For example, a risk of death of 1/1000 would then be valued at ECU 1 million/1000, or 1000 ECU using the VOSL approach. Within a small range of the risk of death at which the VOSL is established this may not be a bad assumption, but it is clearly indefensible for risk levels very different from the one used in obtaining the original estimate.

Estimates of the WTP for a reduction in risk or the WTA of an increase in risk have been made by three methods. First, there are studies that look at the increased compensation individuals need, other things being equal, to work in occupations where the risk of death at work is higher. This provides an estimate of the WTA.

Second, there are studies based on the CVM method, where individuals are questioned about their WTP and WTA for measures that reduce the risk of death from certain activities (e.g., driving); or their WTA for measures that, conceivabl

Table 12.1 European (mainly UK) empirical estimates of the Value of a Statistical Life (adapted from Pearce *et al*, 1992).

Country	Study	Year	ECU million (1995)
Wage Risk			
UK	Melinek	74	0.8
UK	Veljanovski	78	8.4 - 11.8
UK	Needleman	79	0.4
UK	Marin and Psacharopoulos	82	3.7 - 4.2
Average Wage Risk			3.4 - 4.3
CVM Studies[1]			
UK	Melinek	73	0.5
UK	Jones-Lee	76	15.5 - 19.2
UK	Maclean	79	5.2
UK	Frankel	79	5.2 - 21.0
UK	Jones-Lee	85	1.3 - 5.8
SWE	Persson	89	2.6 - 3.2
AU	Maier	89	3.2
Average CVM			4.7 - 8.3
Consumer Market Studies			
UK	Melinek	74	0.4 - 0.8
UK	Ghosh	75	0.8
UK	Jones-Lee	77	1.0 - 11.0
UK	Blomquist	79	1.0 - 3.5
Average UK Market			0.8 - 4.1

1. Includes contingent ranking method.

Table 12.2 Summary table for Value of Statistical Life (adapted from Pearce *et al*, 1992).

	European	USA
	ECU Million. (1995)	
Wage-Risk	3.4 - 4.3	4.2 - 6.6
CVM	4.7 - 8.3	1.7 - 3.0
Market	1.0 - 3.5	1.2 - 1.3
Average	2.5 - 4.4	2.4 - 3.6

12.2.1 Validity of Methods of Estimating the Value of a Statistical Life

All three methods of valuing a statistical life have been subject to considerable criticism. The wage-risk method relies on the assumption that there is enough labour mobility to permit individuals to choose their occupations to reflect all their preferences, one of which is the preference for a level of risk. In economies suffering from long standing structural imbalances in the labour markets this is at best a questionable assumption. Second, it is difficult to distinguish between risks of mortality and morbidity. Third, the WTA will depend on perceived probabilities of death. Almost all studies, however, use a measure of the long-run frequency of death as a measure of risk. This makes the results quoted unsatisfactory. Fourth, the probabilities for which the risks are measured are generally higher than those faced in most of the fuel cycle impacts. This point is returned to below, but a related factor is that the high risk occupations involve individuals whose WTA for an increase in the risk of death is not typical of the population at large (e.g., steeplejacks)[12]. The net impact of all these factors is difficult to gauge but it is *likely* that the estimated WTA will be lower.

12.2.2 Voluntary and Involuntary Risk

There is strong evidence to suggest that individuals treat voluntary risk differently from involuntary risk, with the WTA for a voluntary risk being much lower than that for an involuntary risk. Starr (1976) has estimated, on a judgmental basis, the difference between the willingness to accept a voluntary increase in risk and an involuntary increase. He finds the latter to be around ten times as high as the former for probabilities of death in the range 10^{-6} to 10^{-7}. For lower probabilities that are typical of the fuel cycles, estimates of the differences are not available. In another study of the difference (Litai, 1980), it has been argued that the difference could be as much as 100 times.

The CVM methods are subject to the criticism that the choices are hypothetical and that individuals are not familiar with the concepts of risk involved. Certainly, there have been serious difficulties in conveying the impact of different probability changes through questionnaire methods.

Finally, the consumer expenditure approach is subject to the difficulties that perceived probabilities are very different from objective probabilities, and that the effects of the expenditures are to reduce the risk of death as well as of illness following an accident. It is difficult to separate the two impacts in the studies.

For all these reasons the studies are likely to be biased, with the wage-risk studies producing values that are too low and the CVM studies values that are too high. Taking an average, as has been done here, is averaging unknown errors and one cannot say what the final impact of these errors will be. However, they are all that is available and one can draw some comfort from the fact that the values are, in broad terms, in a plausible range.

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12. This is probably one reason that the estimated value of life declines as the mean risk level in a group increases. From a theoretical perspective one would expect the opposite if the populations were homogeneous. The US Fuel Cycle Report cites data to show that the value of a statistical life more than doubles if one allows for self-selection in the wage risk studies.

This issue is of great importance to the ExternE study, where the public opposition to nuclear fuel will not be reflected in the costs as estimated by using a value of life of around ECU 3.1 million. This may be an issue of perceived versus objective probabilities, but can only be a partial explanation. At this stage, there is no alternative to using this value. However, as part of the ongoing research the issue of voluntary risk needs to be addressed and a revised estimate of the value of life obtained.

12.2.3 Valuation in Different Countries

Within the EU the decision was taken to use a single value for mortality impacts. Although there are differences between the richest and poorest countries in the EU (Denmark at the top is about 60% higher in real per capita GNP than Portugal at the bottom), differences between countries in terms of attitude to risk are not clearly enough identified in the VOSL literature for us to be able to make separate country estimates of VOSL within the EU. For estimates relevant to developing countries, however, we should adjust the values of VOSL. The method proposed is to use estimates based on multiplying the above VOSL by the ratio of the real GNP in the country to the real GNP of the EU[13]. Estimates of this ratio can be made from World Bank data and this has been done for the countries for which data are available. The use of this method was first proposed by Markandya (1994) and has subsequently been adopted by the World Bank and other researchers working in this field. Until we get a better idea of the "GDP elasticity" of VOSL these are the best estimates.

Using these values in the estimation of climate change damages is very controversial. The issue has been raised in the IPCC and national governments of developing countries have not accepted that different VOSLs should apply based on *per capita* income. There is a case to be made for taking an average value for the whole world for such calculations and taking the values given in Table 12.3 for damages that are not related to global damages, such as climate change. If a single global value is to be taken, world actual weighted PPP *per capita* income was $5870 in 1994, including the EU (for countries for which we have data -- i.e. those presented in Table 12.3, plus the EU). The corresponding VOSL figure is 1,026,000 ECU. Note that this value would have to be used for Climate Change mortality impacts, including those in the EU.

13. Also referred to as Purchasing Power Parity (PPP) Adjusted GNP.

Table 12.3a Value of statistical life for countries outside the EU

COUNTRY	PPP GNP US$ 1994	VOSL ECU000 1995	COUNTRY	PPP GNP US$ 1994	VOSL ECU000 1995
ARGENTINA	8720	1524	MALAWI	650	114
ARMENIA	2160	377	MALAYSIA	8440	1475
AUSTRALIA	18120	3166	MALI	520	91
AZERBAIJAN	1510	264	MAURITANIA	1570	274
BANGLADESH	1330	232	MAURITIUS	12720	2222
BELARUS	4320	755	MEXICO	7040	1230
BENIN	1630	285	MOROCCO	3470	606
BOLIVIA	2400	419	MOZAMBIQUE	860	150
BOTSWANA	5210	910	NAMIBIA	4320	755
BRAZIL	5400	943	NEPAL	1230	215
BULGARIA	4380	765	NEW ZEALAND	15870	2773
BURKINA FASO	800	140	NICARAGUA	1800	314
BURUNDI	700	122	NIGER	770	135
CAMEROON	1950	341	NIGERIA	1190	208
CANADA	19960	3487	NORWAY	20210	3531
CEN. AFR. REP.	1160	203	OMAN	8590	1501
CHAD	720	126	PAKISTAN	2130	372
CHILE	8890	1553	PANAMA	5730	1001
CHINA	2510	439	PAPUA NEW GUINEA	2680	468
COLOMBIA	5330	931	PARAGUAY	3550	620
CZECH REPUBLIC	8900	1555	PERU	3610	631
DOMINICAN REP.	3760	657	PHILIPPINES	2740	479
ECUADOR	4190	732	POLAND	5480	957
EGYPT	3720	650	ROMANIA	4090	715
EL SALVADOR	2410	421	RUSSIAN FED	4610	805
ESTONIA	4510	788	RWANDA	330	58
ETHIOPIA	430	75	SAUDI ARABIA	9480	1656
GAMBIA	1100	192	SENEGAL	1580	276
GHANA	2050	358	SIERRA LEONE	700	122
GUATEMALA	3440	601	SINGAPORE	21900	3826
GUINEA-BISSAU	820	143	SLOVENIA	6230	1088
HAITI	930	162	SOUTH AFRICA	5130	896
HONDURAS	1940	339	SRI LANKA	3160	552
HUNGARY	6080	1062	SWITZERLAND	25150	4394
INDIA	1280	224	TAJIKISTAN	970	169
INDONESIA	3600	629	TANZANIA	620	108
ISRAEL	15300	2673	THAILAND	6970	1218
JAMAICA	3400	594	TOGO	1130	197
JAPAN	21140	3693	TRINIDAD & TOBAGO	8670	1515

Table 12.3a (continued).

COUNTRY	PPP GNP US$ 1994	VOSL ECU000 1995	COUNTRY	PPP GNP US$ 1994	VOSL ECU000 1995
JORDAN	4100	716	TUNISIA	5020	877
KAZAKSTAN	2810	491	TURKEY	4710	823
KENYA	1310	229	UGANDA	1410	246
KOREA	10330	1805	UKRAINE	2620	458
KUWAIT	24730	4321	URUGUAY	7710	1347
KYRGYZ REPUBLIC	1730	302	USA	25880	4522
LATVIA	3220	563	UZBEKISTAN	2370	414
LESOTHO	1730	302	VENEZUELA	7770	1358
LITHUANIA	3290	575	ZAMBIA	860	150
MADAGASCAR	640	112	ZIMBABWE	2040	356

SOURCE: WORLD BANK AND EUROSTAT

Table 12.3b Value of statistical life for countries outside EU - Countries for which there are no data

COUNTRY	PPP GNP US$ 1994	VOSL ECU000, 1995
ALBANIA	N.A.	N.A.
ALGERIA	N.A.	N.A.
COSTA RICA	N.A.	N.A.
CROATIA	N.A.	N.A.
GABON	N.A.	N.A.
GEORGIA	N.A.	N.A.
GUINEA	N.A.	N.A.
HONG KONG	N.A.	N.A.
IRAN	N.A.	N.A.
LAO PDR	N.A.	N.A.
MACEDONIA	N.A.	N.A.
MOLDOVIA	N.A.	N.A.
MONGOLIA	N.A.	N.A.
MYANMAR	N.A.	N.A.
SLOVAK REPUBLIC	N.A.	N.A.
TURKMENISTAN	N.A.	N.A.
UNITED ARAB EMIR	N.A.	N.A.
VIETNAM	N.A.	N.A.
YEMEN REPUBLIC	N.A.	N.A.

12.2.4 Allowance for Real Growth

The question has been raised as to whether countries VOSL should be increased in future years to account for *per capita* growth. Within the EU, there was 1.3% growth in *per capita* income over the period 1980-1993. There are no credible projections for the next ten years. Some assumptions have been made for the purposes of modelling climate change impacts in the scenarios proposed by IPCC, though these cover a wide range of 'possible futures' with no clear view on exactly what rate of growth is likely in future years. For consistency, it may be desirable to work with these assumptions, but it is also important to question them. Within the EU, as an alternative, it would be interesting to take a growth rate of around one percent per annum for periods of up to 25 years. For longer periods than that it is probably most appropriate to look at constant *per capita* incomes. The same applies to other OECD countries (with the exception of new members such as Mexico and Poland, which should be treated as described below).

For developing countries and transition economies that are in the growth phase, or will likely move into the growth phase, it is a different matter. If China's GNP is growing at around 8 percent in *per capita* terms, this will alter the VOSL within a short span of time. For such countries, VOSL should be increased in line with GDP growth. The latter should be based on macroeconomic projections that have a wide international consensus of experts.

12.2.5 Transfer of Risk Estimates from Different Probability Ranges

There is the issue of the probability range over which the estimation is carried out and over which it is applied. Typically one is dealing with much lower probabilities of death from the fuel cycle (of the order of 10^{-6} and lower), whereas the studies on which the estimated value of a statistical life is based are dealing with probabilities of between 10^{-1} to 10^{-5}. Furthermore, as the survey by Fisher, Chestnut and Violette (1989) has pointed out, the results from studies at the higher end of the probability range are less reliable. As mentioned earlier, theoretical models would tend to predict that the WTA for lower risks should be lower but if anything, the empirical literature shows the opposite. Partly this is due to the fact that the groups are not homogeneous. The issue remains unresolved and there is little that can be done about this problem at this stage. In the medium term, research on the theoretical and empirical aspects of the problem is needed.

12.2.6 Age Dependent Mortality

The problem of age dependent mortality arises because the value of 3.1 MECU is based on studies in which the individuals involved came from relatively narrow age bands (around 25-55 years, with a concentration in the 35-45 age band). One would expect the VOSL to vary with age, with a possibly lower value for older people, but this does not appear to be supported by the little evidence that is available. Variations of estimated VOSL by age do exist but they do not demonstrate a clear pattern. The one study that provides clear evidence on age dependence is Jones-Lee *et al* (1985), whose results are displayed in Table 12.4. Other studies, such as Shephard and Zeckhauser (1982) address this question but in a purely theoretical context. Jones-Lee *et al* found that VOSL at age 20 is about 70% of VOSL at age 40. Between 20 and 40 it rises slightly, and between 55 and 75 it falls slightly, so that by the age of 75 it is around 77% of the value at age 40.

Table 12.4 Variation in VOSL by age (Jones-Lee *et al*, 1985).

Age	20	25	30	35	40	45	50	55	60	65	70	75
VOSL as % of age 40	68	79	88	95	100	103	104	102	99	94	86	77

These are quite small adjustments to VOSL and are, generally, within the margins of error of the estimates anyway. Furthermore, in those cases where we are going to use VOSL, there are other factors that are more important in determining the value that should be applied than this small age-dependence variation (such as latency, manner of death, years of life remaining etc.). For all these reasons adjusting VOSL for reason of age dependent mortality is not recommended.

12.2.7 Value of Life Years Lost

Some ExternE results on mortality relate to years of life lost rather than increased probability of death. It is possible to estimate the value of a year of life lost from the estimates of the value of a statistical life, if one has data on the age of the reference group, and some way of estimating the discount factor applied to present versus future years of life. Some work along these lines is cited in the US Fuel Cycle Study (Harrison, 1990), and indicates that the implied value of a year of life lost, when used to value average years of life lost from premature death caused by cancer, cardiovascular disease etc., is only around 30% of the value that would emerge from an application of the value of a statistical life.

Pursuing this line of reasoning, one can take a 'prime age male' with 37 years of life expectancy and apply the 3.1 MECU VOSL calculated above. This gives a Value of Life Year Lost (VOLY) for such a person of 70,000 ECU. Years of life lost in the case of premature deaths from chronic diseases such as cancer or cardiovascular disease will vary with age of onset and nature of the condition; but an average estimate of 10-15 years life lost may be reasonable. Taking 12.5 years as the mean of that range would give a total value of 0.9 MECU against a VOSL of 3.1 MECU. The argument is even more compelling for acute effects, where the dose response function is picking the impacts of increases in deaths in the days following higher pollution levels. These are generally thought to be affecting persons with short remaining life expectancy. Hence the use of a VOSL, derived from studies of individuals with normal life expectancies must be questioned there.

What is needed is a framework in which one can use estimates of the value of life years lost, in a way that is consistent with the VOSL values reported above. In the next section such a framework is developed. It is fairly general, has been widely discussed, and can form the basis of adjustments to the valuation of some mortality impacts. Ideally values would be derived from original, well-targeted valuation studies. However, until these are carried out the method adopted here provides useful first estimates that we believe will give a far more reliable answer than the use of the VOSL.

12.3 Mortality and the Value of Life Years Lost (VOLY): Chronic and Acute Effects

12.3.1 Acute Effects

Discussion within the project team on chronic and acute effects related to air pollution tentatively concluded that for acute effects the loss of life years for those affected is *on average* likely to about 0.75 and for chronic effects it is around 5. This is another area that requires further investigation. The relationship between VOSL and VOLY is taken as follows:

$$VOSL_a = VOLY_r \cdot \sum_{i=a+1}^{T} {}_aP_i (1+r)^{i-a-1} \qquad (12.1)$$

Where a is the age of the person whose VOSL is being estimated, ${}_aP_i$ is the conditional probability of survival up to year i, having survived to year *a*. T is the upper age bound and r is the discount rate. The above formula assumes that VOLY is independent of age. This assumption is discussed further below.[14.]

Estimates of survival probabilities for the EU population are available from Eurostat. The male probabilities have been used to estimate VOLY for individuals aged from 35 to 45. The survival probabilities for all ages and age cohorts for the EU are given in Table 12.5. The rate of discount has been taken in the range 0 to 3 %. Assuming that VOSL is 3.1 MECU, the corresponding average value of VOLY for males' aged between 35 and 45 is 84,000 ECU with a 0% discount rate and age 35, to 322,392 ECU with a 10% discount rate and an age of 45. These are given in Table 12.6 below.

Table 12.5 VOLY for different discount rates (ECU, 1995). Corresponding value of VOSL is 3.14 MECU.

Discount Rate	VOLY with Age 35	VOLY with Age 45
0%	84,100	111,600
3%	141,100	168,500
10%	301,400	322,400

14. Note that the formula is set up to allow for the possibility that VSL is age dependent. Given information on the ages of individuals affected by air pollution the estimates of costs can be adjusted for this factor as well, but this has not been done for this analysis.

Table 12.6 Age cohorts and survival probabilities for males in the EU (EUROSTAT, 1995).

AGE OF COHORT	% IN POP	SURVIVAL PROBABILITY	AGE OF COHORT	% IN POP	SURVIVAL PROBABILITY
1	1.176	0.9985	51	1.191	0.9923
2	1.176	0.9988	52	1.191	0.9915
3	1.176	0.9991	53	1.191	0.9906
4	1.176	0.9994	54	1.191	0.9897
5	1.176	0.9998	55	1.191	0.9888
6	1.176	0.9997	56	1.191	0.9872
7	1.176	0.9997	57	1.191	0.9856
8	1.176	0.9997	58	1.191	0.9840
9	1.176	0.9997	59	1.191	0.9823
10	1.176	0.9997	60	1.191	0.9807
11	1.176	0.9996	61	0.974	0.9789
12	1.176	0.9995	62	0.974	0.9771
13	1.176	0.9995	63	0.974	0.9753
14	1.176	0.9994	64	0.974	0.9735
15	1.176	0.9993	65	0.974	0.9717
16	1.176	0.9993	66	0.974	0.9680
17	1.176	0.9992	67	0.974	0.9644
18	1.176	0.9991	68	0.974	0.9608
19	1.176	0.9991	69	0.974	0.9572
20	1.176	0.9990	70	0.974	0.9536
21	1.480	0.9990	71	0.974	0.9485
22	1.480	0.9989	72	0.974	0.9434
23	1.480	0.9989	73	0.974	0.9383
24	1.480	0.9988	74	0.974	0.9332
25	1.480	0.9988	75	0.974	0.9281
26	1.480	0.9987	76	0.974	0.9189
27	1.480	0.9986	77	0.974	0.9097
28	1.480	0.9986	78	0.974	0.9005
29	1.480	0.9985	79	0.974	0.8913
30	1.480	0.9985	80	0.178	0.8821
31	1.480	0.9984	81	0.178	0.8691
32	1.480	0.9983	82	0.178	0.8562
33	1.480	0.9982	83	0.178	0.8432
34	1.480	0.9982	84	0.178	0.8302
35	1.480	0.9981	85	0.178	0.8172
36	1.480	0.9979	86	0.178	0.8001
37	1.480	0.9977	87	0.178	0.7831
38	1.480	0.9975	88	0.178	0.7660
39	1.480	0.9974	89	0.178	0.7489
40	1.480	0.9972	90	0.178	0.7318
41	1.191	0.9969	91	0.178	0.6926
42	1.191	0.9966	92	0.178	0.6534
43	1.191	0.9962	93	0.178	0.6143
44	1.191	0.9959	94	0.178	0.5751
45	1.191	0.9956	95	0.178	0.5359
46	1.191	0.9951	96	0.178	0.4287
47	1.191	0.9946	97	0.178	0.3215
48	1.191	0.9942	98	0.178	0.2144
49	1.191	0.9937	99	0.178	0.1072
50	1.191	0.9932			

Note that the figures in Table 12.5 are the conditional survival probabilities of survival to year i+1, having survived to year i, not the conditional survival probabilities from year a to year i. For someone of age 35 the conditional probabilities are given as:

${}_{35}P_{36}$ = (0.9981). ${}_{36}P_{37}$ = (0.9981)(0.9979) = 0.9960. ${}_{37}P_{38}$ = (0.9981)(0.9979)(0.9977) = 0.9937
and so on.

12.3.1 Empirical Evidence on VOLY and Independence of VOLY from Age

It is important to compare the value of VOLY above with some direct estimates of this variable. The above analysis has assumed that VOLY is independent of age. This is not critical but it makes the analysis a lot easier. The empirical evidence on the value of VOLY is limited. The one study that has focused on this is Moore and Viscusi (1988). Their methodology is similar to that proposed here and assumes a VOLY that is independent of age. With a VOSL of $6.5 million they derive implicit values of an additional year of life of between $170,000 and $200,000. That makes the VOLY about 2.6% to 3.1% of VOSL. In our case the ratio of VOLY to VOSL is between 2.6% and 10.2%, with the higher ratio values corresponding to the 10% discount rate and the lower values to the 0% discount rate. Moore and Viscusi derive VOLY from econometric estimates in which expected life years lost appear as a specific variable. The latter are constructed on the basis of the risk of death and the discounted life years remaining. The two methods are not directly comparable but have the same flavour and, encouragingly, come up with similar answers in terms of the relationship between VOSL and VOLY.

It is recommended to use equation (12.1) to estimate the costs of air pollution via mortality for all cases, using the values of VOLY as given in Table 12.6 and replacing the normal conditional survival probabilities with the probabilities associated with the particular case. For acute effects this calculation is relatively simple. Assuming that individuals are affected immediately, and life expectancy is reduced to 0.75 years, the P_is are replaced with 1 for the following 0.75 years and zero thereafter. This results in an estimated mortality cost of: 73,500 ECU (0% discount rate), 116,250 ECU (3% discount rate) and 234,000 ECU (10% discount rate). Note, however, that the exposure-response function for chronic effects of air pollution implicitly covers reduced life expectancy from acute effects as well as chronic. Investigation of the sensitivities of different approaches suggests that quantification based solely on the recommendations made below for chronic effects will give an acceptably accurate result.

12.3.2 Chronic Effects

For chronic effects the calculation is more complicated. The model followed here is as follows. Once exposed, impacts can occur with a latency that is variable, but could be as much as 50 years. Once the impact is underway, survival probabilities are altered to an extent that is undetermined. Although there are great uncertainties, the study team has estimated, for representative populations, the number of years of life lost as a result of an increment in the hazard in year i, in each future year. Call this $YOLL_i$. We have also estimated for such a case the total number of years of life lost in the population ($YOLL_{tot}$). The value to be attached to a case of chronic mortality can then be estimated as $VOLY_{chronic}$. The formula proposed for this is:

$$VOLY^{r}_{chronic} = \sum_{i=1}^{i=T} \frac{YOLL_i}{YOLL_{tot}} * \frac{VOLY^r}{(1+r)^{i-1}} \quad (12.2)$$

There will be different values for chronic illnesses, depending on the values obtained for $YOLL_i$ and $YOLL_{tot}$, and of course on the value of r. Equation 12.2 has been used to estimate the values for different chronic effects. The results are reported in Table 12.7 below. The age distribution of the population is that of the European Union and the survival probabilities are those for Germany. Note that the last case, where latency and risk are spread out evenly over 30 years is the one considered appropriate for chronic mortality arising from airborne particulate matter. This is discussed below.

A similar analysis has been done for a number of cancers. The figures are given in Table 12.8 below.

In this work estimates were made of latency, years of life lost and survival time after diagnosis. The method by which survival probabilities are estimated is a matter of some discussion. It is, however, outside the scope of the valuation framework as such to go into any further detail. It should also be noted that these figures are being revised, as more accurate data on these epidemiological parameters is being included.

Table 12.7. $VOLY^{r}_{chronic}$ for different latencies and discount rates.

		Discount rate	
	0%	3%	10%
$VOLY^r$	98,000	155,000	312,000
Latency 0 years			
female	98,000	122,123	169,198
male	98,000	120,187	162,059
Latency 30 years			
female	98,000	53,820	11,175
male	98,000	53,810	11,133
Latency and risk distributed over 30 yrs			
female	98,000	84,680	61,269
male	98,000	83,969	59,371

Table 12.8 Estimates of mean years of life lost (YOLL) and subsequent valuation of fatalities, in ECU, for cases of different types of cancer. The cost of illness (estimated at 450,000 ECU) should be added to these values (see Table 12.9).

Type of Cancer/ Discount Rate	Leukaemia	Lung cancer	Stomach cancer	Nasal cancer
Causative pollutants	benzene, butadiene	PAHs, diesel particulates	ethylene oxide	formaldehyde
Latency (l) in years	8	15	15	20
Estimated mean YOLL	22	16	15	14
0% (VOLY= 98,000)	2,160,000	1,570,000	1,470,000	1,370,000
3% (VOLY=155,000)	1,810,000	1,080,000	992,000	848,000
10% (VOLY=312,000)	1,180,000	418,000	373,000	252,000

For cases with a long latency period it is appropriate to add a cost for the period of pain and suffering in addition to the values of life years lost that have been reported above. This is dealt with under morbidity values, in the next section.

Finally, there has been considerable discussion on the valuation of chronic impacts in the dose response functions by Pope *et al*. The issue is that the equations provide an estimate of the years of life lost not the number of excess deaths. As far as the valuation is concerned, we should value years as described in equation 2. The relevant valuations for chronic mortality are those in Table 12.7 in which the latency and risk are uniformly distributed over 30 years. In Table 12.7 the figures are given separately for males and females. Taking a weighted average of 51% female and 49% male, the numbers derived for chronic mortality in the case of PM exposure are: 98,000 ECU (0% discount rate), 84,330 ECU (3% discount rate), and 60,340 ECU (10% discount rate).

12.3.3 Use of the Value of Statistical Life for Mortality Related to Air Pollution

It is recognised that the valuation of mortality is the subject of vigorous debate at the present time. In view of this the use of the VOSL for valuation of air pollution related mortality was retained within the study as part of our sensitivity analysis. The following calculations should be made:

1. Estimates of the number of fatal cancers are given directly by the exposure-response functions.
2. Likewise, estimates of the number of deaths from acute effects of air pollution are given directly by the exposure-response functions.
3. For chronic effects the result from the exposure-response function needs adjustment. First multiply acute deaths by 1.25 to give years of life lost, then subtract this from the output of the function for chronic effects. Divide this result by 5 to give numbers of deaths from chronic effects.

12.3.4 Valuation of Mortality in Accidents and Other Areas here VOSL is appropriate

For the valuation of mortality in accidents we have used the VOSL at 3.1 MECU. This will differ from the value used in transport planning in some countries (in the UK it is now around £1 million or 1.42 MECU). For consistency with the other valuations, however, it is appropriate that we take 3.1 MECU. In general, all situations where the death is premature and applies on average to individuals aged between 25 and 45 the recommendation is that the value 3.1 MECU be used.

If, however, data are available on the loss of life years, and on the gestation period before which the disease is fatal, we can estimate a modified figure based on equation 12.1, with the survival probabilities modified appropriately. Essentially we calculate a new VOSL, given the new P_is and the value of VOLY, which is taken from Table 12.6.

Valuation of morbidity or injury resulting from accidents is discussed in Section 12.4 below.

12.4 Morbidity Impacts

There is an enormous US literature on valuing morbidity effects, and a virtual absence of one in Europe. Given the collaborative nature of this study, the maximum use has been made of the excellent work carried out in this area by the US team, with modifications to their findings as and when appropriate.

The WTP for an illness is composed of the following parts: the value of the time lost because of the illness, the value of the lost utility because of the pain and suffering, and the costs of any expenditures on averting and/or mitigating the effects of the illness. The last category includes both expenditures on prophylactics, as well as on the treatment of the illness once it has occurred. To value these components researchers have estimated the costs of illness, and used CVM methods as well as models of avertive behaviour.

The costs of illness (COI) are the easiest to measure, based either on the actual expenditures associated with different illnesses, or on the expected frequency of the use of different services for different illnesses. The costs of lost time are typically valued at the post-tax wage rate (for the work time lost), and at the opportunity cost of leisure (for the leisure time lost). Typically the latter is between one half and one third of the post-tax wage. Complications arise when the worker can work but is not performing at his full capacity. In that case an estimate of the productivity loss has to be made. It is important to note that COI is only a component of the total cost. However, since the other components are difficult to measure, estimates have been made of the relationship between the total WTP and COI. A COI approach has been used to value non-fatal cancers. The COI plus foregone earnings were the only estimate available. Although not ideal, it is better than the alternative of using no value at all.

CVM is the only approach that can estimate the value of the pain and suffering. The difficulties are those generally associated with the use of CVM and, in addition, of allowing for the fact that it is difficult to know which of the many costs are included in the given responses. In general, respondents will not include those costs that are not borne by them as a result of the illness (e.g., medical insurance). In that event, such costs need to be added. In this category one should also include the cost, in terms of pain and suffering, that the illness causes to other people (the so-called altruistic cost).

Avertive behaviour is the most complex of the three to model. It involves the estimation of a health production function, from which one would be able to estimate the inputs used by the individual in different health states, and taking the difference in value between these obtain the cost of moving from one health state to another. The difficulty is in estimating that function, where many 'inputs' provide more than one service (e.g., bottled water, air conditioners), and where the changes in consumption as a function of the state of illness are difficult to estimate.

The broad groups under which the estimates can be classified are as follows:
a) estimation of restrictive activity days;
b) cost of chronic illnesses;
c) valuation of symptom days;
d) estimation of altruistic costs.

Restricted Activity Days (RADs)
A large number of studies, using COI as well as CVM methods, have been used to estimate several categories of RADs. These are differentiated by illness (respiratory RAD or RRAD, angina RAD etc.), and by severity of impact (minor RAD (MRAD) versus 'normal' RAD). It is stated that these impacts are among the easier of the health impacts to value, as they relate to acute events, lasting a well defined period. The US study provides central or best estimates for these impacts which can, as a first approximation, be taken in the European study using a purchasing power parity (PPP) exchange rate. Although there may be grounds for arguing that medical costs are somewhat higher in the US (at the PPP rate), the errors involved in transferring the estimates are likely to be dwarfed by those arising from other sources. Until a corpus of European studies is available therefore, it is recommended that these values for RAD be taken from the US study. This gives a value of ECU 62 per RAD.

The other alternative is to take a COI approach for the country concerned, and to gross-up the value to get the total WTP by using the estimated factor of between 2.0 and 3.0. The COI can be valued in terms of the medical costs plus any loss of value of time. This approach will have to be taken for those impacts for which the US study does not provide an estimate of the cost of a RAD. As a cross check against the use of US values, this calculation should also be carried out, at least for some of the cases.

The US study (ORNL/RfF, 1994) points out that the central values provided for an RAD cannot be simply multiplied by the number of days lost, because one would expect that the value of each additional day declines as one loses more days, and indeed the empirical evidence supports that (the average value declines as the number of days lost increases). The conclusion that the estimated rate of decline in the value of an RAD with the number of days is too inaccurate to be of use is probably correct, making the use of a single value the best course to follow at this stage.

Chronic Illness
The valuation of the chronic illness is largely in terms of the COI approach (although there are a few CVM studies). The COI approach includes the direct as well as the indirect costs of the illness (such as lost earnings and loss of leisure time). The CVM approach operates in terms similar to the value of statistical life (VOSL) - i.e. by asking what the willingness to pay to reduce the risk of contracting a chronic respiratory illness would be. The corresponding value is referred to as the 'value of a statistical case' (or VSC). The US study quotes the Krupnick and Cropper, (1992) study which shows that the best estimate for the VSC for respiratory disease is around US$1 million (0.9 MECU 1995). Given that a central estimate of the VOSL *in the US study* is around US$2.5 million (2.3 MECU 1995), this would suggest that the value of a chronic VSC is around 40% of that of the corresponding VOSL. This can only be an indicative figure but, as a first guideline to the values in this area, it is probably not too bad. In due course more detailed costs should be computed and compared with those that emerge from the use of the 0.4 * VOSL valuation (in conjunction with the relevant probabilities)[15].

One area where this kind of valuation arises is in relation to the severe hereditary effects associated with some nuclear fuel cycle impacts. Here, it was felt that the severity of the event, and the seriousness of the public's perception of it, warranted the use of a value as high as that for a statistical life, i.e. 3.1 MECU. Chronic morbidity impacts for chronic bronchitis in adults were also quantified in relation to particulate air pollution in the fossil fuel cycles. The exposure-response function used gave extra cases at a point in time, rather than extra new cases annually. Attributing a full VSC to the extra (as distinct from the new) cases each year would have seriously over-valued the impacts. In practice, these have been valued as for an acute episode of bronchitis annually, which is in effect, under-valuing these impacts. Some further work is necessary. Chronic morbidity impacts also arise occupationally (see below).

Symptom-Days
The US Fuel Cycle Study reviews the extensive literature on the valuation of symptom-days. These include CVM studies, as well as some that combine avertive behaviour and CVM. Although the work carried out is impressive, there are unfortunately, still many difficulties to be resolved. The CVM studies have problems of low response rates and extreme bids that have to be discounted. There is also a difficulty in knowing the extent to which the responses include the use of avertive measures. Some of the results appear to indicate that the latter are not always allowed for. Some recent studies (Dickie *et al,* 1986, 1987) have included information on actual

15. The COI approach can be quite complex, including the impacts of pollutants on earnings now and in the future. A case of the latter is the impact of lead on children's IQ, which thereby effects their future earnings potential. In this context it is important not to double count; if expenditures on compensatory education are included, then the costs of lower IQs cannot (or only to the extent that the compensatory education is ineffective).

avertive behaviour and revealed responses but, as the US Fuel Cycle Study points out, ‘the results of this study need considerable refinement before they can be used with confidence in a morbidity benefit analysis. The limitations arise in the theory, data, statistical, and implementation phases of the study’.

The most recent study of this type in Europe was provided by Navrud (1997). The author performed a CV study of a representative sample of 1000 Norwegians (above 15 years) to estimate their WTP to avoid 1 and 14 additional days annually (subsample A and B respectively) of seven ‘light’ health symptoms and asthma. the seven light symptoms were: coughing, sinus congestion, throat irritation, eye irritation, headache, shortness of breath and acute bronchitis. First, people were asked about how many days they experienced having each of these symptoms in the last 12 months; then they were asked what costs this implied in terms of medicines, hospital visits and lost time at school, work or leisure. Finally, they were asked about their maximum WTP to avoid having 1 or 14 more days of each symptom the next year, compared to what they had experienced last year.

Air pollution was not mentioned in the survey; thus the values obtained are noncontextual. The numbers are believed to be more transferable from one type of project to another, and possibly between countries but do not relate specifically to air pollution problems.

In general, however, the state of the art is not yet at a stage where symptom-days can be valued with any confidence (even by the less rigorous standards that apply to environmental valuation as a whole). Nevertheless, given the availability of one plausible value, it is better than excluding the endpoint altogether, and a value of ECU 7.5 (1995 prices) per symptom day (see ORNL/RfF, 1994) has been included in the analysis. In the valuation by Navrud of a symptom day, covering the range of symptoms described above, he gets values of 2.5 ECU to 7.5 ECU. The value of a symptom day in ExternE (1995 prices) is 7.5 ECU, which is at the upper end of the range from Navrud.

Altruistic Impacts

As with symptom-days, estimates of the impact of an illness on the utility of others is not at a stage at which it can be used in a valuation exercise. One US study (Viscusi, Magat and Forrest, 1988) came up with an altruistic value for each case of poisoning avoided of more than 5 times the private valuation. The experiment consisted of a CVM in which individuals were asked their WTP for a TV campaign that would reduce poisoning resulting from poor handling of insecticides. However, the study had a relatively unsophisticated design and the results need to be confirmed in other studies. Work in the UK by Needleman (1976) and Jones-Lee *et al* (1985) has suggested that the altruistic values are around 40-50% of the private total valuations. Again, however, these are isolated findings and need to be corroborated.

In view of the current state of the art in this area, altruistic valuations have not so far been included in the ExternE study. However, the omission of altruism provides a bias to underestimation of damages.

There are a number of endpoints that have been valued. The most recent European evidence is from Navrud (1997), with Rowe *et al* (1995) providing a review of the more recent US studies.

Chest Discomfort. Based on CVM studies, a central value of 7.5 ECU was taken.

Emergency Room Visits (ERV). The WTP to avoid an ERV is estimated at 223 ECU. This has been taken to value ERVs generated as a result of concentrations of PM_{10} and ozone.

Respiratory Hospital Admissions (RHA). The WTP to avoid an RHA is estimated at 7,870 ECU. This value has been taken to value ERVs generated as a result of concentrations of PM_{10} and ozone.

Children's Bronchitis

Table 12.9. Valuation of morbidity endpoints in the ExternE Study,

ENDPOINT	VALUE (ECU)	ESTIMATION METHOD AND COMMENTS
Acute Morbidity		
Restricted Activity Day (RAD)	75	CVM in US estimating WTP. Inflation adjustment made.
Symptom Day (SD) and Minor Restricted Activity Day	7.5	CVM in US estimating WTP. Account has been taken of Navrud's study. Inflation adjustment made
Chest Discomfort Day or Acute Effect in Asthmatics (Wheeze)	7.5	CVM in US estimating WTP. Same value applies to children and adults. Inflation adjustment made.
Emergency Room Visits (ERV)	223	CVM in US estimating WTP. Inflation adjustment made.
Respiratory Hospital Admissions (RHA)	7,870	CVM in US estimating WTP. Inflation adjustment made.
Cardiovascular Hospital Admissions	7,870	As above. Inflation adjustment made.
Acute Asthma Attack	37	COI (adjusted to allow for difference between COI and WTP). Applies to both children and adults. Inflation adjustment made.
Chronic Morbidity		
Chronic Illness (VSC)	1,200,000	CVM in US estimating WTP. Inflation adjustment made
Chronic Bronchitis in Adults	105,000	Rowe *et al* (1995).
Non fatal Cancer	450,000	US study revised for inflation.
Malignant Neoplasms	450,000	AM suggested valuing as non-fatal cancer.
Chronic Case of Asthma	105,000	Based on treating chronic asthma as new cases of chronic bronchitis.
Cases of change in prevalence of bronchitis in children	225	Treated as cases of acute bronchitis.
Cases of change in prevalence of cough in children	225	As above.
Other Effects		
Severe Hereditary Effect	3,140,000	Perception of severity of effect discussed in original ExternE. Inflation adjustment made.
Occupational Injuries (minor)	77.5	French compensation payments, increased for inflation.
Occupational Injuries (major)	22,600	French compensation payments increased for inflation.
Workers & Public Accidents (minor)	6,970	TRL (1995). New estimates.
Workers & Public Accidents (major)	95,050	TRL (1995). New estimates.

Occupational morbidity and mortality impacts
The valuation of morbidity impacts for workers generally takes the same form as that for the lay public[16]. There are, however, some exceptions and complications. One is the case of injuries, which is dealt with separately below. The other is the treatment of illnesses that involve compensation for pain and suffering. In part this makes up for loss of earnings and in part it is recompense for pain and suffering. However, it is difficult to disentangle the two. Hence it is not recommended that these values be used, other than in the case of accidents. Finally there are some morbidity impacts that do not have values from the general literature. For these an initial valuation has to be made.

In the coal fuel cycle, the following impacts were evaluated:
- coal miners' mortality from exposure to radon and lung cancer;
- coal miners' morbidity from simple pneumoconiosis (CWSP) and from its advanced form (progressive massive fibrosis (PMF));
- respiratory symptoms in coal miners.

The valuation of mortality is straightforward in the case of additional deaths from lung cancer and exposure to radon. These are valued using VOSL. Briefly, CWSP is in itself not necessarily disabling and so, at this point, has not been valued. CWSP increases the risk of developing PMF which is disabling and is associated with increased mortality. The risks of developing PMF have been estimated directly. For valuation purposes, only the effects of PMF on premature deaths have been considered, by estimating at 2%, the proportion of those miners who get PMF who also subsequently die of pneumoconiosis as cause of death.

For morbidity impacts such as cases of breathlessness and cases of phlegm and chronic coughing without breathlessness a valuation procedure similar to that for the general public is appropriate.

It should be noted that there are other possible mining-related impacts that have not been quantified because of a lack of data, for example:
- mortality from non-malignant lung disease other than PMF
- possible other cancers, such as lung cancer from quartz dust and diesel, or digestive cancer
- back pain and other musculo-skeletal problems

12.4.1 Accidents

The final category of impacts concerns accidents during the various activities of the fuel cycle. Accidents occur both to those employed in the energy industry, as well as to the general public. The activities that have to be analyzed with regard to the coal fuel cycle, for example, are:
- coal mining
- coal transportation
- power station construction
- power station operation
- power station related road traffic

16. The interpretation to be put on the value will, however, depend on what is assumed about the internalization of the external effect.

For those that are directly employed in the sector, the key issue is the extent to which such impacts are internalized. The decision within ExternE is to value such impacts and then present the figures separately from those of the costs that clearly fall into the external costs category. For deaths, the recommended VOSL should be used. For injuries the costs should be based on the costs of illness, plus any compensation that is payable for that accident, treating the latter as a proxy for the pain and suffering. This is not ideal but is the best available in the circumstances.

Values for accidents are available from the UK Department of Transport. A 'serious' injury is valued by the Department of Transport at £15,000 (21,000 ECU), and a 'minor' injury at around £300 (420 ECU). These figures are fairly arbitrary being derived, if anything, from a consideration of COI plus the awards of the injuries boards. Nevertheless they provide some indication of the values. Pearce *et al* (1992), have suggested that 'speculatively' an upper bound might be 3400 ECU for minor injuries and 200,000 ECU for major ones.

None of these values are based on the willingness to pay approach. In contrast various US estimates of accidents can be referred to, which are based on a WTP approach. Two studies have been identified as important. These are Moore and Viscusi (1988) and Martinello and Meng (1992: see ORNL/RfF, 1994). In addition, there is one UK study also based on WTP measures (TRL, 1995). All three are discussed here.

In the ORNL/RFF report the best study (i.e. with a WTP basis) was Moore and Viscusi. They present a range of estimates for WTP to avoid a non-fatal injury of 15,900 ECU to 31,800 ECU (converted from the original to 1995 prices). Their best estimate is 24,300 ECU. The figure applies as an average for all injuries in a sample where the baseline risk of workdays lost was 4.7 per 100 working days.

A less theoretically rigorous but more recent study is by Martinello and Meng (1992: see ORNL/RfF, 1994). They use a 1986 database, the sample being blue-collar workers in logging, mining and manufacturing industries. The occupational injury rate averaged 6.3 days per 100 in this sample. The model does not include worker compensation adjustment, uses pre-tax wage rates and is estimated using OLS. Given these problems one would expect the aggregate injury rates to be lower than other studies, which they are. Their range of estimates is equivalent to ECU 7480 - 9350.

Martinello and Meng also distinguish between major and minor injuries. They found that wage compensation was insignificant (actually negative and significant) for minor injuries but was positive, significant and large for major injuries. The mean estimate here is equivalent to 118,800 ECU.

A WTP approach to the valuation of road accidents was used by the UK by the Transport Research Laboratory (TRL). Non-fatal injuries were classified according to severity (either serious or slight) and to improve the accuracy of the results these groups were subdivided into more homogenous subgroups according to the extent and duration of pain, period of hospital treatment, recovery time and level of residual disability. Cost estimates for these subgroups were weighted together to derive average costs for casualties with serious or slight injuries. Their results are shown in Table 12.10. It can be seen that the average figures are strikingly similar to the best estimate given by Moore and Viscusi of 24,300 ECU.

Table 12.10. Accident Costs (1995 ECU) Estimated by TRL

Casualty Severity	Human Costs	Medical & Support	Total
Serious	86,550	8,500	95,050
Slight	6,335	635	6,970
Average	25,330	1,795	27,125

In conclusion, the values in the TRL study are credible and consistent with those obtained from the US study. Hence they should be used to update the earlier estimates of accidents, for both serious and slight categories.

In estimating the accidents related to the fuel cycles, it is necessary to look at accident statistics. For example, in the UK coal fuel cycle study the categories of accident measured conformed to the UK Reporting of Injuries Diseases and Dangerous Occurrences Regulations (RIDDOR):

- Fatalities
- Major injuries, including amputation, major fractures, serious eye injuries, loss of consciousness and any injuries requiring hospital treatment over 24 hours.
- Minor injuries including other accidents responsible for the loss of more than three working days.

It is important to note that this valuation of health impacts discussed so far relates to ‘routine’ operations of the plants and does not include the valuation of disasters. The difficulties that arise with valuing the latter are primarily in terms of defining reasonable probabilities of the events. The issues that arise are extremely complex and are discussed separately in Chapter 7, and by Markandya (1998).

12.5 Conclusions

This Chapter has reviewed the literature on the valuation of the health impacts of the fuel cycles. Health impacts are probably the most important of all to value and also the most difficult conceptually. The Chapter began by looking at the methodological issues arising in the valuation. Impacts to be valued were divided into mortality, morbidity and accidents.

For mortality impacts there are several issues that need to be resolved but, at the present time, a value of 3.1 MECU in 1995 prices appears to be the best central estimate. A range of problems that need to be addressed in the future was identified. VOSL should not be used in cases where the hazard has a significant latency period before impact, or where the probability of survival after impact is altered over a prolonged period. In such cases the value of life years lost (VOLY) is recommended. However, in view of the continuing debate in this area VOSL should be retained for sensitivity analysis.

For morbidity, categories of impacts were divided into estimates of restrictive activity days (RADs) and variants of that; estimates of chronic illness, symptom-days, and altruistic impacts. For RADs and variants, it was suggested that where US estimates were available, these should be used as one source, with conversion into ECU made using a purchasing power parity exchange rate. In addition, it was also suggested that a cost of illness (COI) approach be used, with the COI value being grossed-up for items not captured by that method (i.e., pain and suffering). In some cases the latter approach will be the only one possible as there are no estimates of RAD available. An estimate of the grossing-up factor was provided. Estimates of chronic illness can be made via two routes. One is to use the COI and gross-up as indicated above. Another is to use estimate of the value of statistical case, as estimated in the US and convert it in the manner indicated above. For symptom-days only a first attempt of the estimates exists. The same applied for altruistic impacts.

A distinction was made between occupational and general health impacts. For workers there are some illnesses that need to be valued individually. However, for the majority of them the same valuation as is used for the general public is appropriate. There are also accidents that workers suffer that differ in kind from the general public. For non-fatal accidents, data on compensation (in conjunction with COI) is probably the best at this stage. For fatal injuries, the estimated value of a statistical life should be employed.

In the fossil fuel cycles, impacts have been estimated for incremental PM_{10} pollution and, where pollution modeling permits, for incremental ozone also. The PM_{10} and O_3 impacts are taken as additive. No direct effects of SO_2 or NO_x are estimated; but they both contribute indirectly via the formation of secondary particulates, while NOx is also a precursor of ozone. For implementation, the estimated impacts have been linked up with economic valuation as described in the present chapter.

Finally the Chapter raised the issue of valuing non-routine accidents, as well as some conceptual difficulties that arise particularly in connection with the nuclear fuel cycle. The difficulty here is that the methods discussed above do not capture the extent of public concern. The issue is of sufficient importance to be treated separately and Chapter 7 is devoted to it.

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13. IMPACTS ON TERRESTRIAL ECOSYSTEMS

13.1 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. Treatment of the impacts of global climate change is discussed in Chapter 6 and in more detail in the separate volume on Global Warming Damages (European Commission, 1998). 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 seem less important than the effects of air pollution because they act over a restricted range. Indeed, associated damage may well contravene permits granted to a site operator. They may be significant if an ecosystem of special value (for whatever reason) is damaged, or if an area of land becomes so contaminated as to threaten local sustainability. Consideration of such cases is too specific to particular energy projects to consider in the ExternE Project.

Until recently most pollution problems were thought to be 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 usually limited to a small area (relative to the continental scales that are now of concern in Europe). 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 crops, forests and natural ecosystems at concentrations 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.

For the most part this Chapter is concerned with effects of air pollution on plants. Direct effects of air pollution on animals have rarely been observed and then only at very high concentrations of pollutants. Effects on invertebrates have frequently been observed, though as a secondary effect of impacts on plants, for example through changes in primary productivity or in plant amino acid production.

Section 13.2 reviews the importance of different pollutants in causing damage, and the mechanisms by which damage occurs, with particular emphasis on forest damage. The Section concludes with a review of the impact pathways for agricultural crops, forests and other terrestrial ecosystems. Section 13.3 reviews current thinking on critical loads and levels for the main air pollutants. Sections 13.4 to 13.6 deal with the exposure-response data for ozone, SO_2 and NO_x that are available from the literature. Section 13.7 discusses the effects of interactions between pollutants. Section 13.8 reports on the methodology used for quantifying the damage from acidic deposition to agricultural soils, and Section 13.9 reports on interactions between air pollution and the performance of plant pests and pathogens.

Section 13.10 discusses valuation of the damages identified in the Chapter. Overall Conclusions are then provided. Finally, Appendix 13.1 discusses the different means of reporting ozone concentrations.

13.2 Effects and Mechanisms of Air Pollution on Plants

13.2.1 Relative Role of Different Pollutants

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 13.1. 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 PORG, 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_2 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 (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 13.1 (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 fertilisational effects. For other ecosystems the ranking will vary. Low productivity ecosystems are especially sensitive to nitrogen inputs. The situation is complicated by interactions between pollutants.

Table 13.1 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 SO_2.

Pollutant	**Forests**	**Crops**	**Natural flora**	**Natural fauna**	**Fisheries**
SO_2	XX	XX	XX	X	
NO_x	X	X	X	X	
NH_3	X	X	X	X	
O_3	XXX	XXX	XXX	X	
Total acid	XXX	X	XXX	X	XXX
Total nitrogen	XXX	X	XXX	X	XX

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 larger animals indirectly through effects on plants used for food or shelter. However, performance of many insects, particularly pests, has been found to be influenced by pollution.

13.2.2 Damage Mechanisms

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. Primary impacts on plant productivity,
5. Secondary impacts on (e.g.) competitive status, resistance to pests and disease, at both the species level and the level of the individual plant.

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.

Foliar uptake of SO_2 may lead to several effects, including:

1. Loss of stomatal control (Mansfield and Freer-Smith, 1984);
2. Alterations to pH of cell ed witMotiv be influ5eace of manynthesi72a7.0007 9(witMotivy)28.3(l) Tw[((t

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*, 1993). 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.

13.2.3 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_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_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_3, SO_2 and NO_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_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 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 high pollutant doses, they do show that current ambient pollution levels are capable of affecting plants.

13.2.4 Forest Declines

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, CO_2 uptake and storage, water management and wildlife habitat. Trees are in many ways a fundamental feature of 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 recent 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 SO_2 levels (Knabe, 1970; Farrar *et al*, 1977). SO_2 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 SO_2 applied for 2 hours. However, Shaw *et al* (1993) established a much lower threshold for direct SO_2 damage to needles of highly sensitive individuals in a genetically variable population of Scots pine, finding classic SO_2 induced needle tip necrosis associated with a mean of <10 ppb over a 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 were 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 Calcium (Ca) and Magnesium (Mg). Increased Aluminium (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 12.3 below, 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.

13.2.4.1 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 NO_x or NH_3) 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 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 Ca, Mg, and potassium (K) or phosphorus (P). This damage type has been connected with elevated levels of SO_2 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. Rehfuess (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, Manganese (Mn) and Iron (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 spruce in the coastal plains of Germany, Belgium and the Netherlands are affected by this problem which is believed to be associated with NH_3 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 NH_3 is not emitted from fuel cycles in large quantities, there is a strong interaction between SO_2 and NH_3, 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 broad-leaved 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.

13.2.4.2 Forest health in the USA

NAPAP concluded that the most forests in the US are not affected by decline (Barnard and Lucier, 1991), though a number of declines had been observed there 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 O_3 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. O_3 damage has also been reported in other areas of the western US including the Sierra Nevada and San Gabriel mountains. Miller (1989) discusses these declines in more detail. 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 identified as highly sensitive to O_3 in the 1960s (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. This decline seems to have had little effect on overall growth of sugar maple.

13.2.4.3 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 - SO_2 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 O_3 damage, symptoms are often unequivocal and can be safely ascribed of secondary stresses: interactions between numerous pollutants and insect pests, pathogens, cold stress and drought have been recorded. In some of the pollution induced declines discussed, but not all, there is clear connection between forest condition and productivity.

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-Grerup 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.

13.2.5 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. Pathways describing effects on natural flora and fauna, crops and forests are shown in Figures 13.1 to 13.3. To a large degree these pathways are similar: the main differences arising through valuation.

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) spatiallytruncated to simplify the analysis;
- Uncertainty is assessed for any estimate of damage.

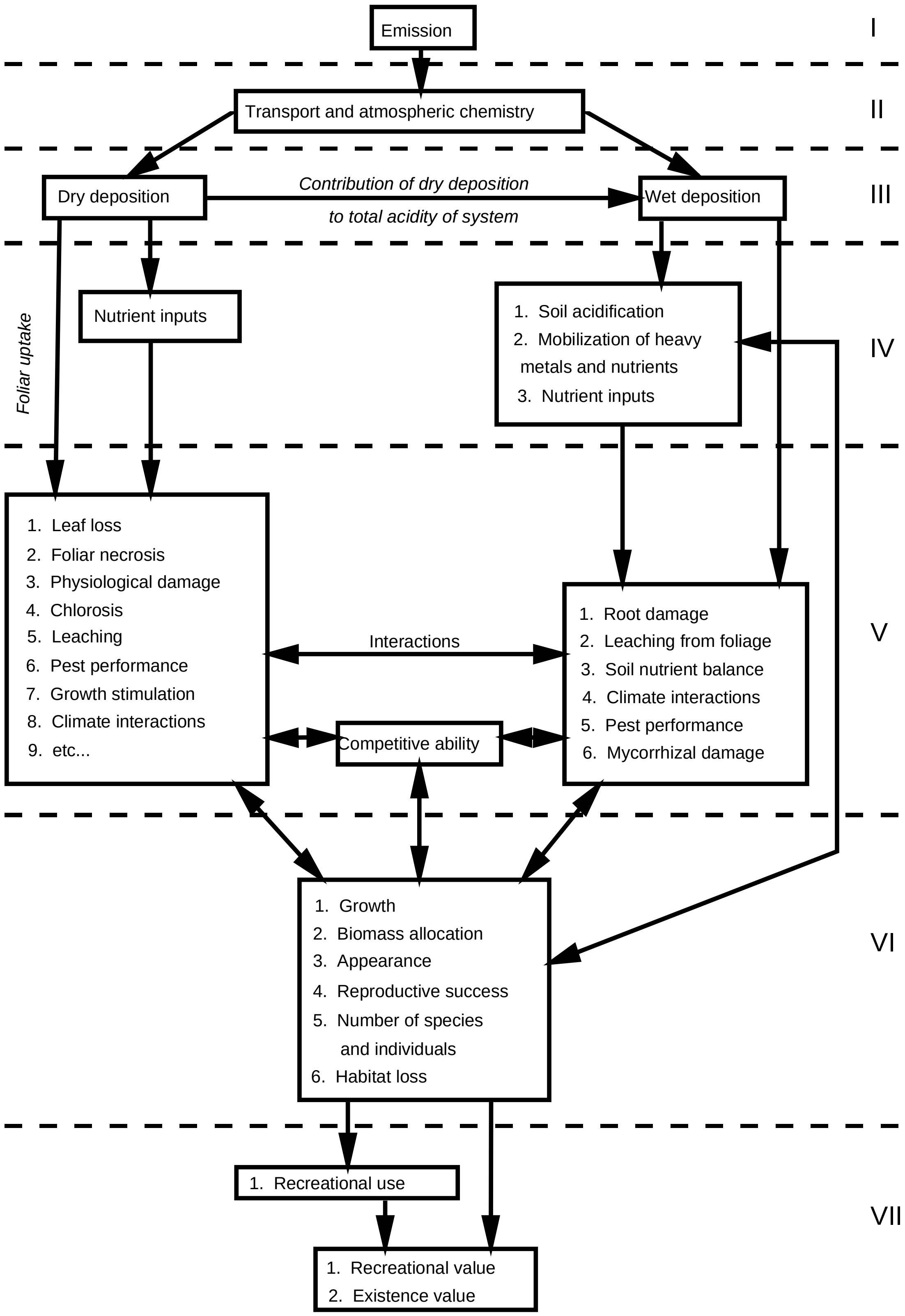

Figure 13.1 An impact pathway typical of those used to describe the effects of acidic deposition and photo-oxidants on natural terrestrial ecosystems. All known effects have been included in this pathway, whether they are quantifiable using current knowledge or not.

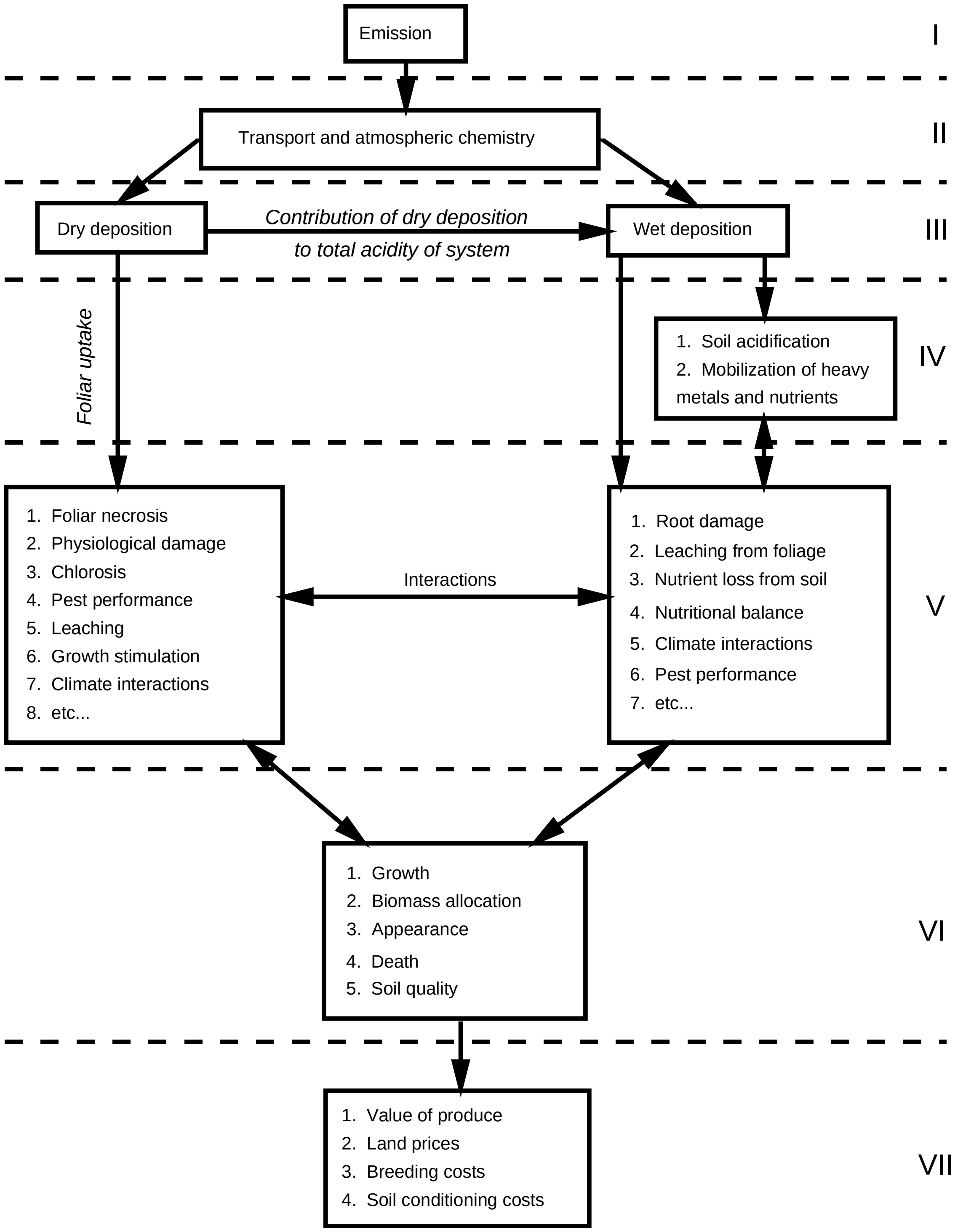

Figure 13.2 Impact pathway illustrating the effects of air pollution on agricultural crops.

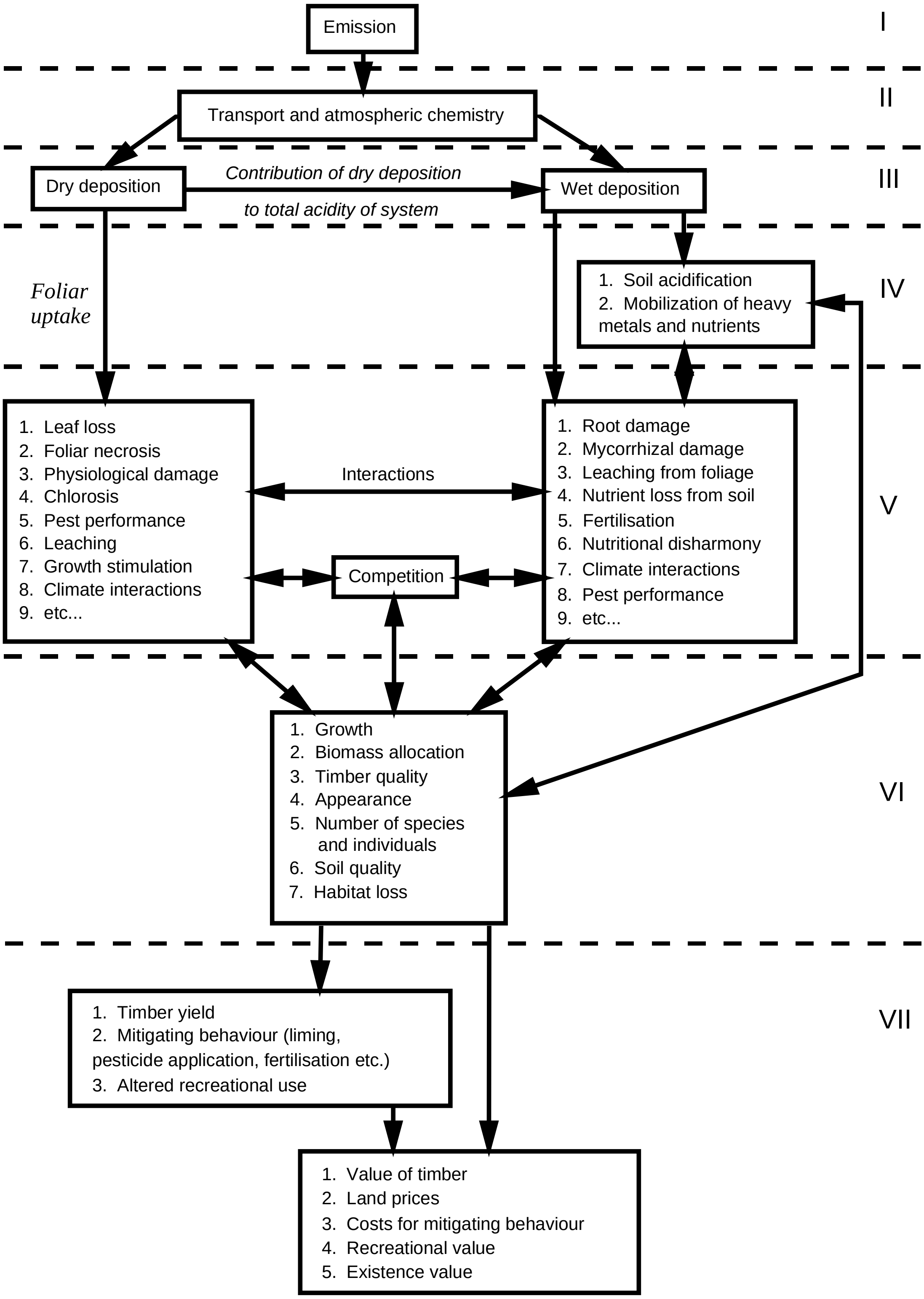

Figure 13.3 Impact pathway for effects of acidic deposition and photo-oxidants on forests.

13.2.6 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;
10. Economic valuation.

A large body 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 im

- Models reviewed by NAPAP;
- The IIASA Forest Study Model;
- The forest module of the RAINS model;
- Sverdrup and Warfvinge (1993);
- Kuylenstierna and Chadwick (1994).

We concluded that none of these permitted reasonable assessment of the damages to forests likely to result from fossil fuel pollution at the scale required here. The approach identified by Kuylenstierna and Chadwick was most favourably reviewed, because it used 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 highlighted the fact that there is benefit to be gained from examination of the wealth of data that is available for assessment of forest damage.

In addition to damage costs it is necessary to consider any expense undertaken to mitigate against damage. Kroth *et al* (1989) assessed the silvicultural measures which forest managers apply to counteract forest damages, and associated costs for Germany. Total costs for West Germany were found to be in the range of 41.2 to 112.9 MECU/year for the five year period from 1988 to 1992. A wide range of measures were employed; site mapping, liming and fertilisation, reduction of deer grazing, reforestation of damaged stands, remodelling, cultivation of underwood, renovation of 'protection forests' and biological protection of forests. The total can be divided by the total area of spruce forest in the former West Germany subject to critical loads or levels exceedence 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 exceedence 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. The analysis is not applicable outside of Germany, because of a lack of information on the application of measures for protecting forests from acidification.

13.3 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 load 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.

13.3.1 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 13.2). 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 13.3). 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.

13.3.2 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 13.4. Values have in most cases been set on the basis of empirical field observations or chamber studies.

Table 13.2 Critical loads of acidity and sulphur for forest soils (compiled from Nilsson and Grennfelt, 1988).

Class	Minerals controlling weathering	Usual parent rock	Total acidity (kmol H^+ km^{-2} $year^{-1}$)	Equivalent of sulphur (kg ha^{-1} $year^{-1}$)
1	Quartz K-feldspar	Granite Quartzite	< 20	< 3
2	Muscovite Plagioclase Biotite (<5%)	Granite Gneiss	20 - 50	3 - 8
3	Biotite Amphibole (<5%)	Granodiorite Greywakee Schist Gabbro	50 - 100	8 - 16
4	Pyroxene Epidote Olivine (<5%)	Gabbro Basalt	100 - 200	16 - 32
5	Carbonates	Limestone Marlstone	> 200	> 32

Table 13.3 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).

Type of ecosystem	Critical load (kg N ha^{-1} yr^{-1})	Reliability	Indication
Acidic (managed) coniferous forest	15-20	Reasonable	Changes in ground flora and mycorrhizal fruit bodies
Acidic (managed) deciduous forest	<15-20	Reasonable	Changes ground flora
Calcareous forests	Unknown[1]	-	Unknown
Acidic (unmanaged) forest	Unknown[1]	-	Unknown
Lowland dry-heathland	15-20	Good	Transition of heather to grass
Lowland wet-heathland	17-22	Good	Transition of heather to grass
Species-rich lowland heaths/ acid grassland	7-20	Reasonable	Decline of sensitive species
Arctic and alpine heaths	5-15	Reasonable	Decline of lichens, mosses and evergreen dwarf shrubs, increase in grasses and herbs
Calcareous species-rich grassland	14-25	Good	Increased tall grasses, reduction of diversity
Neutral-acid species-rich grassland	20-30	Reasonable	Increased tall grasses, reduction of diversity
Montane-subalpine grassland	10-15	Reasonable	Increased tall graminoids, reduction of diversity
Shallow soft-water bodies	5-10	Good	Decline of isoetid species
Mesotrophic fens	20-35	Reasonable	Increased tall graminoids, reduction of diversity
Ombrotrophic bogs	5-10	Reasonable	Decline of typical mosses, increased tall graminoids

Table 13.4. Critical levels of SO_2, NO_x, O_3, and rain and cloud composition by exposure duration and land use.

Pollutant	Land use category	Exposure duration	Level, µg m^{-3} unless otherwise stated
NO_2[1]	All	Annual mean	30
		4 hour mean	95
O_3	All	Annual	300 ppb.hours above 40 ppb baseline
SO_2	Forests	Annual and winter mean	15/20[2]
	Natural vegetation	Annual and winter mean	15/20[2]
	Crops	Annual and winter mean	30
	Cyanobacterial lichens	Annual mean	10
NH_3	All	1 hour	3300
		1 day	270
		1 month	23
		1 year	8
Rain and cloud	Crops	Growing season mean	1 mmol H^+ l^{-1}
	Bryophytes/ lichens	1 hour (single episode)	3 mmol H^+ l^{-1}
		Annual total	600 mol H^+ ha^{-1} yr^{-1}
	Forests [3]	Time weighted annual mean [4]	0.3 mmol (H^+ or NH_4^+) l^{-1} with 0.15mmol SO_4^{2-} l^{-1}
		Annual mean	1 µg S m^{-3} as particulate sulphate

[1] Critical level for NO_2 in combination with SO_2 and O_3 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 NH_4^+ concentrations.
[4] This mean to be applied where ground-level cloud persists for >10% of the time.

13.4 Exposure-Response Functions for Ozone

13.4.1 Background

The long-term critical level of ozone for forest trees and agricultural crops has been expressed as cumulative exposure to a concentration of over 40 ppb, abbreviated to AOT40 (Fuhrer and Achermann, 1994). This is generally regarded as the threshold value above which toxic effects might be anticipated, as few adverse effects have been reported to occur at lower concentrations (Braun and Flückiger, 1995). However, Kärenlampi and Skärby (1996) emphasised that 40 ppb should not be regarded as a lower concentration limit for biological effects, as some biological responses may occur at lower concentrations. Pääkkönen *et al* (1997), for example, showed the best correlation for growth responses and leaf senescence in *Betula pendula* was achieved with an exposure index based on AOT30. AOT40 is calculated as the total hours of exposure x mean concentration (or individual concentration values) above 40 ppb, and is expressed as ppb-h or ppm-h. For herbaceous plants and agricultural crops, it is summed for a three month period, recording ozone concentrations during daylight hours; for trees it is summed over a six-month growing season. Where predicted yield reductions are less than 10%, it is generally recommended that AOT40 should not be used because of difficulties with the degree of resolution (Fuhrer and Achermann, 1994). This recommendation is not followed in the present study: here we believe that it is better to quantify and then to deal with uncertainties in a clear and transparent manner.

The most up-to-date studies on the effects of ozone exposure on a range of tree, crop and natural vegetation types are reported in a range of papers in Kärenlampi and Skärby (1996), published from the United Nations Economic Commission for Europe (UN-ECE) Convention on Long-Range Transboundary Air Pollution in a workshop in Kuopio, Finland in April 1996.

Other data on dose responses, based on different dose measures, have been included in this Chapter, many of which relate to the United States studies during the National Crop Loss Assessment Network programme (NCLAN) in the 1980s. These were yield responses plotted against seasonal mean ozone concentrations, for exposure periods of 7 h per day, later changed to 12 h per day.

13.4.2 Difficulties in Deriving Dose Responses and in Providing Estimates

In UN-ECE, a distinction has been made between so-called Level I and Level II critical levels. At Level I, there is no distinction between plant species, and a single critical level is set, which, in principle, would protect the most sensitive known receptor under the most sensitive environmental conditions. There is some concern that Level I critical levels will not be sufficiently rigorous to protect some of the more sensitive high value crops which have been identified as being particularly susceptible to ozone. These include tobacco, tomato, potato, rape, soybean, watermelon, fruit trees, vines and other perennial crops (Kärenlampi and Skärby, 1996). Level II studies would include more detailed assessment of yield losses against AOT40, i.e. dose response relationships, and would need to take into account a number of other factors, which are listed below. While many of these factors have been identified as important variables in individual studies, there are many problems with defining critical levels at Level II, and few data available which can be used to derive dose response curves.

Some of the problems are:

- Related species may show very different responses, and there may be large differences in sensitivity between different cultivars, or clones, of the same species (e.g. Pääkkönen *et al*, 1995).
- Different stages in the life cycle may exhibit differences in sensitivity e.g. full-grown trees may be more sensitive than seedlings (Kelly *et al*

compared growth of 6 year-old *Fagus* saplings in OTCs with CF and ambient air (NF). AOT40 in NF treatment was 16.2 ppm-h. Early senescence was apparent in the NF trees, which became more pronounced with time. Growth was adversely affected by the third growing season. They comment that the AOT40 value of 10 ppm-h is probably stringent enough for this species. Braun and Flückiger (1994) analysed results from different experiments and concluded that a cumulative dose of 7 ppm-h per year above 40 ppb would result in 10% yield reduction in 3 years, i.e. a total dose of 22 ppm-h. There was a decline in total biomass as a percentage of the control in 7 experiments, which ranged from about 5% at 5 ppm-h to more than 30% at 70 ppm-h. Mortensen *et al* (1995) showed that 14.4 ppm-h resulted in yield loss in seedlings of 10% over a period of 2.5 years. Küppers *et al* (1994) predicted that an ozone dose of 12 ppm-h calculated for 24 h or 10 ppm-h for daylight hours would reduce shoot biomass by 10% in 2 years' fumigation.

13.4.3.2 Fraxinus excelsior

No measured effect at ozone concentrations of 150 nl l^{-1} (ppb) for 8 h-days over 3 years of exposure of between 24 and 27 days per annum (Colls *et al*, 1995). The experimental conditions were severe relative to UK ambient air concentrations. The duration of leaf fall period was increased by exposure to ozone, but other aspects of growth were unaffected. Landholt *et al* (1996) exposed ash seedlings to different ozone concentrations in OTCs for one season. There was no significant effect on biomass at the highest concentration, which was 50% ambient plus 30 ppb at high altitude (1700 m a.s.l).

13.4.3.3 Quercus robur

See comment for *Fagus*. Küppers *et al* (1994) show a similar relationship for oak, with possibly a more severe effect on shoot growth (though the degree of experimental variation was not recorded); a dose of 12 ppm-h resulted in a shoot biomass reduction of 10% after 2 years' fumigation.

13.4.3.4 Quercus rubra (North American species)

A 10% reduction in growth threshold would occur at approximately 32 ppm-h in mature trees, assuming that the 34% reduction in stem growth increment associated with an approximately 10-fold increase in exposure from ambient to twice ambient, is part of a linear response. However, seedling growth was largely unresponsive (Kelly *et al* 1995). Kelting *et al* (1995) showed that a mean concentration of 82 ppb ozone for 7 h reduced mature tree cumulative fine root production and turnover by 33 and 42% respectively compared with growth in ambient air. However, there was no effect on fine root turnover in seedling trees.

13.4.3.5 Betula pendula, B. pubescens and *B. verrucosa*

Two critical levels, one of 9 ppm-h and the other, 13 ppm-h, are given by Fuhrer and Achermann (1994) quoting Matyssek *et al* (1992). The difference between these two is that the first is based on a 1-year experiment. with reference to a control of zero ozone. The second is based on biomass reduction relative to biomass at 20 ppb ozone (reference concentration) which is determined by interpolation between 0 ppb and 50 ppb, assuming a linear response relationship. Pääkkönen *et al* (1995) exposed clonal material at 3 development stages to 12 h daily exposure of 50, 90 and 130 ppb ozone for 25 d. Response varied with the 3 different developmental stages of 2-year old saplings. Height growth response was reduced from 35 cm to 27 cm between AOT40 of 0 and 22, i.e. a 25% reduction at 22 ppm-h in the most sensitive developmental stage. An approximately 10% reduction occurred at about 12 ppm-h in the group of plants that had immature but rapidly expanding leaves. However, in the treatment where the leaves emerged under ozone fumigation, there was a positive effect of increasing AOT40. Five *B. pendula* clones were grown under field conditions in ozone 1.2-1.7 times higher than ambient. Growth responses varied from the most sensitive, which had decreased biomass production at the lowest treatment, to the most tolerant clone with unaffected growth rate (Pääkkönen *et al*, 1995). Mortensen and Skre (1990) studied effect of low ozone concentrations on Betula pubescens, B. verrucosa and Alnus incana seedlings. Fifty days exposure at 25, 35, 53 and 82 ppb for 7 h per day resulted in an almost linear decline in height of the 3 species, from 35 to 20 cm. Visible ozone injury appeared at 53 ppb.

13.4.3.6 Alnus incana

See data from Mortensen and Skre (1990) which compared this species with 2 birch species. There was a linear decline in shoot and root weight between ozone doses of 25 and 82 ppb, although the proportional decline was less than in birch. Leaf senescence was also enhanced.

13.4.3.7 Populus deltoides x nigra

Growth response of rooted cuttings to 3 ozone treatments at 0.5, 1.0 and 2.0 times ambient, showed a non-linear decline in stem height, mass and basal diameter (Woodbury *et al*, 1994).

13.4.3.8 Populus tremuloides

Growth response equations for stem and root biomass of rooted cuttings and seedlings against ozone exposure between 10 and 105 ppm-h showed linear declines for both, particularly root biomass (Karnosky *et al*, 1996). There were genotypic variations in response, and one clone was tolerant. Two clones, one ozone-sensitive and one ozone-tolerant, were grown in ozone or ozone and elevated carbon dioxide (Kull *et al*, 1996). Ozone caused a decline in photosynthetic rate, even in the ozone-tolerant clone. Elevated carbon dioxide exacerbated the detrimental effects.

*13.4.3.9 Eucalyptus (*various species*)*

Monk and Murray (1995) used two treatments of 26 and 172 ppb (7-h mean) for 7 h per day in 5 days out of 14 for 18 weeks. Eight *Eucalyptus* species were fumigated in OTCs at 7 months' of age. Significant differences were found between the species: two (*E. micocorys* and *E. gomphocephala*) showed 30% wt. reduction, although there were no visible signs of leaf injury in the latter species. There was no visible or biomass reduction effect on *E. globulus*. Other species that were adversely affected, but to a lesser degree were *E. viminalis, E. grandis, E. camaldulensis, E .robusta* and *E. marginata*. Various other studies, referred to in the Monk and Murray paper, quote work done on the visual assessment of ozone damage to Eucalypts. In view of the finding that *E. gomphocephala* showed a growth reduction of 30%, without any sign of visible injury, they need to be interpreted with caution.

13.4.3.10 North Eastern American hardwoods

Rebbeck (1996) exposed seedlings of *Prunus serotina*, *Acer saccharum* and *Liriodendron tulipifera* to two seasons of ozone between 16 and 197 ppm-h. There were large differences in response; *Prunus* growth decreased with increasing ozone, *Liriodendron* increased and *Acer* was not significantly affected after one season's growth. After 2 season's exposure, neither *Acer* or *Liriodendron* were affected, but *Prunu*s showed a 32% reduction in total plant biomass at twice ambient ozone concentrations, and 10% at ambient.

13.4.3.11 All broad-leaved species.

A linear regression of percent change in growth against dose (ppm-h) was compiled from data in various listed papers, mostly based on North American species (see Reich, 1987). Yield reduction was approximately 70% at 100ppm-h dose, and 30% at 50 ppm-h.

13.4.4 Exposure-response Functions for Coniferous Trees

13.4.4.1 Picea abies (the most studied of European coniferous tree species).

Wieser *et al* (1996) studied effects of ozone on photosynthesis in mature trees, using a twig chamber system. Only at the highest ozone concentrations of 100 ppb persisting for one vegetation period or more (equivalent to a dose of 100 ppm-h), was there a significant effect on photosynthesis and then only in the current flush of needles. Braun and Flückiger (1995) recorded no effect of ozone under the conditions used for *Fagus sylvatica*. Küppers *et al* (1994) recorded a percentage reduction from control growth of 16% in the shoots at AOT40 of 31 ppm-h after two years' exposure for two year-old seedlings. (For the same dose exposure, they recorded a reduction in beech of 32% and oak of 36%, although no errors were given). Skärby *et al* (1995) showed no measured effect in 3 different concentrations over 5 seasons: CF, ambient and enriched ambient. Photosynthetic rate was adversely affected, but more so in older trees. Davis and Wood (1972) showed that the species was resistant to 250 ppb ozone for 8 hr (see also larch). Slovik *et al* (1996) showed that percentage damage to Norway spruce canopies was positively correlated with SO_2, but not NO_2 or ozone. Reich (1987) regarded Norway spruce as a very ozone-tolerant species. Barnes *et al* (1990) exposed three year-old trees to 200 or 40 (control) µg m^{-3} of ozone with and without acid mist in soils of low and high nutrient status for two years. Ozone and/or acid mist had only minor (<10%:NS) effects on light-saturated photosynthetic rate. Dobson *et al* (1990) conducted experiments in which combinations of ozone exposure and drought were imposed on *P. abies* and *P. sitchensis*. Exposure to ozone did not affect photosynthetic rate or stomatal conductance.

13.4.4.2 Picea sitchensis

Wilhour and Neely (1977) showed a 14% reduction in stem weight in an OTC experiment of 126 days at 100 ppb (76 ppm-h). See also Dobson *et al* (1990), where results were as for *P abies*. Lucas and Diggle (1997) compared growth responses of *P. abies* and *P. sitchensis* after 3 years of ozone exposure. *P. abies* was unaffected, but they calculated a critical level of 21.3 ppm-h over 40 ppb for *P. sitchensis*.

13.4.4.3 Pine (Various species, including N. American ones)

Pinus strobus: Sensitive clones may be injured by ozone at 70 ppb for 4 h (Costonis and Sinclair, 1969 in Krupa and Manning, 1988).
Pinus taeda: Four full-sib families were exposed daily over 3 growing seasons to between 22-92 ppb ozone. Dose response models predicted growth reductions between 0 and 19% after 2 years and 13% in the most sensitive family after 3 years (Shafer and Heagle 1989). Mature trees showed a rapid negative growth response (in 1-3 days) as measured by change in circumference growth at 40 ppb ozone or above (McLaughlin and Downing, 1996). Reinert *et al* (1996) showed that ozone has serious consequences for growth and development of this species. Exposure to 320 ppb for 23 weeks reduced various growth parameters by between 14 and 35%.
Pinus elliottii: Hogsett *et al* (1985) exposed two varieties of slash pine to 104 or 76 ppb ozone as 7 h seasonal means for 112 days. When compared with growth in charcoal filtered air:

Plant height was reduced 41% in var. *densa*
Top dry wt. was reduced 50% in var. *elliotti*
Root dry wt. was reduced 68% in var. *densa*

Pinus sylvestris: Davis and Wood (1972) showed that this species was sensitive to 250 ppb ozone TJ/r0006n3 weeks reduced9 to 250 ppb

13.4.4.5 Larix decidua

Sensitive to 250 ppb for 8 h at 2 week intervals starting 4 weeks after needle emergence. (Davis and Wood, 1972).

13.4.4.6 All coniferous species

A frequent generalisation is that coniferous species seem less sensitive than broad-leaves. However, that is based on similar exposure patterns, so may not hold true if allowance is made for year-round needle exposure. A linear regression of percent change in growth against dose (ppm-h) was compiled from data in various listed papers (see Figure 2a of Reich, 1987).

13.4.5 All Trees

13.4.5.1 Broad leaved trees and conifers.

An AOT40 value of 12 ppm-h accumulated over 3 years was given by Braun and Fluckiger (1994), based on data compiled by Pye (1988). Several Eastern US tree species, as 2-4 wk old seedlings, were exposed to 50, 100 and 150 ppb ozone. Loblolly pine and American sycamore showed reduced growth. White Ash and yellow poplar showed increased growth at 50 ppb (Kress and Skelly, 1982).

13.4.5.2 Forest trees and field crops

A threshold level of 40 ppb above AOT40 was set for crops and forest trees (Pleijel *et al*, 1995). Yield reduction in agricultural crops, forest trees and natural vegetation was highly correlated with AOT40. Critical level to protect trees from growth reductions was set at an AOT40 of 10 ppm accumulated for 24 hours per day over 6-month growing season (Sanders *et al* 1995).

A general comment about tree sensitivities: Reich (1987) in his review concluded that conifers are, at least initially, less sensitive than hardwoods. Data presented by Pye (1988) in his review do not show this pattern. However, poplar can be a particularly sensitive species, and Reich' review contained a number of experimental studies on poplar, which may have distorted the trend. There can be large clonal differences, e.g. Pye refers to Patton (1981) who showed that poplar clones that grew fastest at low ozone levels showed greatest reductions in growth under elevated levels.

13.4.6 Agricultural Cereal Crops

Kärenlampi and Skärby (1996) state that at Level I mapping, the exceedence of the critical level of 3 ppm-h should not be converted into a yield loss estimate, but only used as an indication of the degree of risk. The AOT40 for crops is calculated for a three month period during daylight hours, defined as those hours with a clear sky global radiation of 50 W m-2 or more, and calculated as the highest running three month sum during the period when crops are grown.

13.4.6.1 Wheat (Triticum aestivum)

Analysis of data from open-top chamber experiments demonstrated a linear relationship between relative yield of spring and winter wheat and AOT40 (Fuhrer in Kärenlampi and Skärby, 1996). This is the most complete data compilation to date, and is based on results from 10 seasons, 6 countries and 10 cultivars. The regression shown in Figure 2 gives the AOT40 value corresponding to a 10% loss as 5.7 ppm-h, and a linear equation:

$$y= 99.7 - 1.7 \times O_3 \text{ (ppm-h)}; \quad r2=0.89$$

Statistical analysis (Pleijel *et al*, 1996) showed that the least significant deviation from a 100% yield that can be estimated with 99% confidence is 4-5%, and the critical level for a 5% yield loss is approx. 2.8 ppm-h. A response relation for wheat, based on a regression of relative yield reduction against AOT40 for data compiled from 5 growing seasons, 4 countries and 4 cultivars was earlier given by Fuhrer and Achermann (1994):

Relative yield reduction	AOT40 (ppm-h)
5%	2.6
10%	5.3
15%	7.9
20%	10.5

(The relationship was recommended for use up to 35 ppm-h, but not for yield reductions below 10%). The relationship in Kärenlampi and Skärby (1996) was based on additional data, and slightly increased the AOT40 values at which critical yield reductions (e.g. 5, 10%) occurred. Heagle (1989) in an NCLAN study showed percent loss in yield in statistically significant ozone treatments as a dose response:

Ozone (ppb) seasonal 7h day mean concentration	40	50	60
% loss in yield	8	13	17

(the percent reduction was based on the yield obtained at a seasonal mean of 25 ppb for 7h exposure).

A 33% reduction in yield was shown in NF air at 42 ppb compared with CF at 22 ppb when applied for 60% of the season (Heagle, 1989). A 0.8% crop yield loss was shown with both ambient ozone and 3 ozone standard scenarios (50 and 60 ppb for growing season 12-h means, and all hours > 100 ppb (Olszyk *et al*, 1988). This reduction was smaller than other published effects. Brewer *et al* (1988) predicted losses of 0 - 20% for winter wheat as a result of ambient levels of rural ozone in the Tennessee Valley (seasonal means for 12-h between May and September were 45 to 60 ppb). Brown *et al* (1995) used ITE landclass maps and ozone levels to predict losses of winter wheat with AOT40 values. Soja and Soja (1995) gave results from a closed chamber fumigation experiment to study sensitivity differences during plant development of winter wheat *cv. Perlo.* Ear growth was most affected, then stem growth, and total leaf production. Comparing 78 versus 15 ppb for 8h/day, early growth stages responded with rapid leaf green area loss.

In later stages of development, there was little difference in plant response between control and fumigated treatments.

13.4.6.2 Durum wheat (Triticum durum)

Badiani *et al* (1996) studied the effect of different concentrations of ozone on 2 cultivars of durum wheat, which is an economically important crop in Mediterranean countries. One cultivar was ozone-sensitive and the other ozone-resistant. According to criteria adopted by UN-ECE, a critical ozone level of 5.4 ppm-h AOT40 was calculated for the sensitive cultivar.

13.4.6.3 Maize (Zea mays)

In a review paper of data acquired in NCLAN studies, Heagle (1989) derived a dose response:

Ozone (ppb) seasonal 7h day mean concentration	40	50	60
% loss in yield	1	3	5

There was no effect in NF air at 70 ppb compared with CF at 20 ppb for 80% of the season. (Heagle, 1989). There was no effect in NF air at 44 ppb compared with CF at 15 ppb for 84% of the season (Heagle, 1989). Field corn responded with between 1.1 and 1.5 % yield reductions in 3 scenarios: 50 ppb, 60 ppb, both for seasonal 12-h means, and 100 ppb for all hours (Olszyk *et al*, 1988). Brewer *et al* (1988) predicted losses of between 0 and 6% as a result of ambient levels of rural ozone (see under wheat).

13.4.6.4 Barley (Hordeum vulgare)

There was no effect on yield in the 3 scenarios in 2.2.2.3 above (Olszyk *et al* 1988). Ashmore (1984) showed no effect on growth in non-filtered air as opposed to filtered in 2 cultivars, but a third cultivar did show significant differences between the treatments at developmental stages although not at the final harvest. In this experiment, the mean daily maximum concentration was 51 ppb ozone, while the concentration exceeded 60 ppb on 28% of the days. Fuhrer in Kärenlampi and Skärby (1996) showed a dose response for barley in Figure 9:

$$y = 99.71 - 1.519 \times O_3 \text{ (ppm-h)}; \qquad r2=0.97$$

This value is similar to the slope for wheat (-1.7), which suggests that sensitivity to ozone in barley is similar to that for wheat.

13.4.6.5 Sorghum bicolor

Heagle (1989) showed only a small loss in yield:

Ozone (ppb) seasonal 7h day mean concentration	40	50	60
% loss in yield	1	2	3

13.4.6.6 Cereal crops in general

Skärby *et al* (1996) reviewed the results of 32 experiments carried out in the European Open-Top Chamber Programme. Results indicated that ambient ozone levels in Europe are high enough to reduce grain yield of spring wheat. Grain yield of barley and oats was less affected. There is little information on how climate influences the response of crop plants to ozone, but several of the experiments reviewed were carried out over more than one growing season, during which there were large differences in weather conditions. Response of the plants to ozone was, nevertheless, very similar.

13.4.7 Pasture Grasses and Clover

13.4.7.1 Lolium perenne

Ashenden *et al* (1996) showed a substantial yield reduction of 37.1% in an ozone treatment of 40 ppb, with 2 peaks of 3 h at 80 ppb followed by 1 h at 100 ppb after 18 weeks' exposure.

13.4.7.2 Agrostis capillaris

The treatment listed above in Section 13.4.7.2 did not affect the yield of this species (Ashenden *et al*, 1996).

13.4.7.3 Clover species

Heck *et al* (1986) reviewed NCLAN studies and summarised the following findings: 300 and 600 ppb ozone for 2 hours, applied twice, 7 days apart at 4 stages of development. Shoot and root growth was affected and nodulation decreased. 300, 500 and 800 ppb for 7 hours per day for 6 months. Total forage and forage regrowth was reduced in clover and clover-fescue mixture. 30, 50 and 90 ppb ozone applied for 7 hours per day during the growing season over 2 years. Forage regrowth reduced by 14% and 27% in two higher ozone treatments in the second year. There was a 22% and 54% reduction in clover at the higher levels and some increase in fescue as clover was lost, also a reduction in forage quality. 50, 100 and 150 ppb ozone was applied for 4 hrs per day for 6 days per week for 32 days. Maximum recorded shoot reduction was 24%.

13.4.7.4 Trifolium repens.

It appears to be the least sensitive to ozone of the clovers, although there was a range of responses in five different cultivars. The most sensitive were *L. sacramento* and *L. californica*. Generally there was visible leaf injury at ambient concentrations (Becker *et al*, 1989). They state that studies are required to determine whether visual damage is accompanied by yield reduction. Other results based on *T. repens* can be summarised as follows: Heagle *et al* (1996) showed that 2 clones, one of which was sensitive (NC-S) and the other resistant (NC-R), had different responses when grown in non-filtered air compared with CF air in a greenhouse or an open-top chamber. A yield reduction of more than 50% was recorded by the third successive harvest in the more sensitive cultivar. Mortensen and Bastrup-Birk (1996) exposed the cultivar Menna to 4 levels of ozone: CF, NF, NF + 25 ppb and NF + 50 ppb for 8 weeks at 8 hours per day. Half the plants were protected with ethylene diurea (EDU). Growth was significantly reduced with increasing ozone dose, and stolon production was more affected than leaves. The threshold at which no effects were detected was between 20 and 40 ppb. The AOT40 value for a 10% reduction of biomass was 4.7 ppm-h, similar to, but slightly lower than that for wheat (5.7 ppm-h). Fuhrer in Kärenlampi and Skärby (1996) showed a dose response relationship, of dry weight ratio between EDU-treated plants and untreated plants against ppm-h AOT40. The critical level for a 10% reduction was about 6.8 ppm-h and the equation:

$$y = 1.0 - .0156 \times O_3 \quad r2=0.449$$

13.4.7.5 Mixed pasture species.

Ashmore and Ainsworth (1995) exposed pots of mixtures of *Agrostis capillaris*, *Festuca rubra* and the forbs, *Trifolium repens* and *Veronica chamaedrys* to ozone in OTCs. Total ozone exposure ranged from 800 ppb above 40 ppb to 15000 ppb above 40 ppb. Half the pots were cut at 2-week intervals. Linear relationships were fitted between biomass or proportion of each species and ozone exposure. Whether cut or uncut, *T. repens* biomass declined with higher levels of ozone, but *F. rubra* and *A. capillaris* increased, so total biomass remained similar. Generally the adverse effects on the forbs was greater in the cut treatments. Pleijel *et al* (1996) exposed a field-grown grass-clover mixture (Trifolium pratense, Phleum pratense and Festuca pratensis) to 4 ozone treatments in OTCs and to ambient air outside, for 2 consecutive growing seasons. There was a significant negative relationship between yield and ozone concentration, so that yield at an AOT40 of 45 ppm-h was about 4% less than in CF air. The clover component declined significantly in the second growing season.

13.4.7.6 Mixed forage species (American)

Johnson *et al* (1996) exposed a forage mixture of alfalfa (*Medicago sativa*) and timothy (*Phleum pratense*) to two CO_2 concentrations and four weekly ozone episodes of 8-h duration with peak daily concentrations of 30, 80, 130 and 180 ppb. Increasing ozone concentration decreased alfalfa shoot biomass at the higher CO_2 concentration only. Timothy showed reduced shoot growth at the intermediate ozone levels, but the highest level of ozone resulted in more shoot growth in the mixture at both CO_2 levels. Heagle (1989) listed the dose responses of mixed grass and legume forage:

	% yield reduction at different ozone levels		
Ozone (ppb)	40	50	60
Festuca arundinacea and Trifolium repens	6	10	15
Phleum pratense and T. pratense	9	19	31

The latter mixture showed the greatest reduction in yield at both 50 and 60 ppb ozone of any of the crops listed in the table.

13.4.8 Agricultural (Vegetable and Fruit) Crops

13.4.8.1 Solanum tuberosum (potato)

Heagle (1989) showed a 17% yield reduction in NF air at 44 ppb compared with CF (CF, 21 ppb) for 81 % of time from plant emergence to maturity. Heagle (1989), reporting on a different experiment, showed a 14% yield reduction in NF air at 48 ppb, compared with CF at 27 ppb, for 73% exposure time. Heagle (1989) gives a dose response that is similar to that shown by wheat (18% loss at 60 ppb):

Ozone (ppb) seasonal 7h day mean concentration	40	50	60
% loss in yield	9	14	19

Heck *et al* (1986) summarised other experiments: With three different cultivars, EDU treatment in ambient ozone showed that exposure to ozone resulted in a 25% reduction in cultivar, 'Norland' and 31% reduction in 'Norchip'. With two different cultivars, ozone exposure at 200 ppb for 3 hours at several growth stages decreased tuber weight.

13.4.8.2 Glycine maxima (soybean)

Heagle and Letchworth (1982) reported a yield reduction of 8-25% of the control in 9 cultivars (a tenth one was not affected) when ozone was added to non-filtered air to produce a seasonal 7 h per day mean of 100 ppb between May and October. Yield reduction was in comparison with growth in filtered air. Heck *et al* (1986) showed that when ozone was added at 22 and 112 ppb for 7 hrs per day, there was a 39% reduction in yield at the higher ozone treatment and a 12.6% reduction in seed oil. Heck *et al* (1984) described which measure gave the best dose response relationship. A mean ozone exposure statistic was better than peak ozone values for predicting yield loss. Cure *et al* (1986) used different growth yield models to predict reductions. Models based on mean ozone values were better and less variable than ones based on peak values or ranges (confirmed findings in Heck *et al*, 1984). Heagle *et al* (1987) applied a 7h mean ozone concentration of 27 ppb for 2 yr (109 days) and compared yield with that in CF air. The predicted yield loss at ambient ozone concentrations of 54 ppb was 12% in 1983 and 14% in 1984. Brewer *et al* (1988) used crop loss models developed under NCLAN to estimate potential yield loss from ambient ozone concentrations monitored in the Tennessee Valley. Predicted losses ranged from 6 to 23% compared with yields at 25 ppb ozone. Heagle (1989) calculated a dose response:

Ozone (ppb) seasonal 7h day mean concentration	40	50	60
% loss in yield	5	10	16

These yield losses are similar to, but slightly lower than those of wheat at the same concentrations are.

13.4.8.3 Phaseolus vulgaris (kidney bean)

Salam and Soja (1995) demonstrated that 18 ppm-h was required before reduction in photosynthetic activity occurred. Like many legumes, *P. vulgaris* is very sensitive to ozone.

The most sensitive cultivar showed a 14% reduction in yield in ambient concentrations of ozone (Moser *et al* 1988). A short-term critical level for injury development: AOT40 of 0.7 ppm-h accumulated over 3 consecutive days is given by Fuhrer and Achermann (1994) but needs validating. This value is also used for other sensitive crops such as clover. Various responses were shown in an experiment repeated 12 times. In three experiments, there was a significant reduction of up to 20% in 53 ppb ozone in NF air compared with 25 ppb ozone for 76% of the season (Heagle, 1989). Up to 27% reduction in yield was measured in 52 ppb NF air compared with 23 ppb in CF for 75% of the season (Heagle, 1989). In dry beans, a yield reduction ranging from 17.5 to 23.5% in 4 different scenarios including ambient ozone: 50, 60 ppb for 12-h days etc was measured (see Table 13.5 for a reproduction of the data of Olszyk *et al*, 1988).

13.4.8.4 Various types of bean

Heck *et al* (1986) showed growth responses under a range of conditions: EDU-treated white beans in ambient ozone: plants treated with EDU showed yield increases of 0 - 37% in 3 cultivars. 30 and 60 ppb ozone applied for 1.5 h twice to 6 growth stages in snap bean reduced relative and absolute growth rates. Variable concentrations and duration: short-term changes in allocation to root in snap bean; no yield effect on resistant cultivar, persistent allocation changes in sensitive one. Ambient ozone in snap beans: 41% increase with benomyl spray. Colls *et al* (1993) exposed two species of bean, *Phaseolus vulgaris* and *Vicia faba* to CF ambient air or to elevated ozone concentrations. They proposed that a 5% yield reduction would be a criterion for significant change, and that a reference ozone concentration of 10 ppb could be used, on the basis that the background European concentration has risen from 10 to 20 ppb in recent decades. Then the ozone concentration to cause a 5% yield reduction is 25 ppb. If the ozone concentration were reduced to 10 ppb from 30, then Phaseolus could be expected to show an 8% yield increase, and 15% if grown at 40 ppb. Vandermeiren and De Temmerman (1996) exposed two cultivars of *Phaseolus* to a sub-acute fumigation of 100 ppb ozone for 7h per day for 8-9 days before or after flowering, or to 70 ppb for both periods. Both treatments resulted in an AOT40 of 3.33 ppm-h. Plants in CF air were used as controls. The effect of exposure was much more severe during the reproductive stage, leading to an 11% reduction in pod yield as fresh weight, or when exposure was over both periods, leading to a 20% fresh weight reduction. Thus, yield effects cannot be explained solely by the ozone dose. Heagle (1989) calculated a dose response:

Ozone (ppb) seasonal 7h day mean concentration	40	50	60
% loss in yield	4	9	15

13.4.8.5 Lycopersicon esculentum (tomato)

Heagle (1989) calculated % yield loss against dose:

Ozone (ppb) seasonal 7h day mean concentration	40	50	60
% loss in yield	5	10	18

Olszyk *et al* (1988) showed a 2.9-4.2% reduction in yield with 4 different scenarios, including 50 and 60 ppb ozone and ambient US concentrations. Heck *et al* (1986) summarised experimental results: 80 to 100 ppb ozone for 5 h, 5 days per week for 5 weeks resulted in an 86% reduction in numbers of fruit, and a 91% reduction in fruit weight in the variety, Tiny Tim. 400 ppb, applied to 4 cultivars for 2 hours six times in the seedling stage, caused a 57% reduction in marketable yield of the most sensitive cultivar. In an ambient ozone gradient, a multiple regression was derived which predicted that ozone was responsible for 85% of reduced fruit size along the gradient. The model predicted a 50% yield reduction at an ozone dose of 20 ppm-h for >100 ppb. The tomato cultivar was '6718 VF'.

13.4.8.6 Grapes

Olszyk *et al* (1988) listed a 15.9 - 22.4% reduction in yield with 4 different scenarios . Losses were predicted to be the highest at ambient US concentrations.

13.4.8.7 Lemons

Olszyk *et al* (1988) listed a 17.7 -20.4% reduction in yield with 4 different scenarios. Losses were predicted to be the highest at ambient US concentrations.

13.4.8.8 Oranges

Olsyk *et al* (1988) listed a 14.8 -19.8% reduction in yield with 4 different scenarios. Losses were predicted to be the highest at ambient US concentrations.

13.4.8.9 Onions

Olszyk *et al* (1988) listed a 21.4 - 24.4% yield reduction with 4 different scenarios. Losses were predicted to be the highest at ambient US concentrations.

13.4.8.10 Lettuce

Olszyk *et al* (1988) showed no yield reduction with 4 different scenarios. Heagle (1989) also found no effect on yield.

13.4.8.11 Spinach

Olszyk *et al* (1988) recorded a 3.5% yield reduction with 4 different scenarios.

13.4.8.12 Strawberries

Olszyk *et al* (1988) recorded no yield reduction with 4 different scenarios. However Kentgen et al (1997) showed that some cultivar are more tolerant then others due to differences in sensitivity of the guard cells in the stomata. 'Elsanta', which is most widely planted in Germany, is the most tolerant cultivar.

13.4.8.13 Sugar beet

Olszyk *et al* (1988) recorded no yield reduction with 4 different scenarios.

13.4.8.14 Pepper

Heck *et al* (1986) reported an experiment with 120 or 200 ppb ozone for 3 hours 3 times per week. There was a 16% and 54% loss in total fruit dry weight and a 77% loss in mature fruit in 200 ppb.

13.4.8.15 Carrot

Heck *et al* (1986) reported results with 190 or 250 ppb ozone for 6 hrs, 1.5 exposures per week. The ozone doses caused a 32% and 46% decrease in root dry matter respectively.

13.4.8.16 Parsley

Heck *et al* (1986) reported results with 200 ppb ozone for 4 h twice a week for 8 weeks, which resulted in a 23% decrease in plant dry weight. The greatest effect was at the initial exposure.

13.4.8.17 Citrullus lanatus (watermelon)

Eason *et al* (1996) grew three cultivars in CF air with 5 levels of SO_2 in the presence (80 ppb) or absence of ozone for 4 hours per day, 5 days per week, for 22 days. In the presence of ozone, SO_2 increased foliar injury in all 3 cultivars, but the most serious effect was in most ozone sensitive cultivar. Thus there is a synergistic effect of the two air pollutants on this crop. Reinert *et al* (1992) showed that foliar damage evident in some areas of cultivation in the Ebro delta (Spain) could be reproduced in OTCs in which plants were exposed to non-filtered or CF air. A scale of damage indicated that most leaves were affected in ambient air, but non-filtered air in the OTCs also caused much more damage than charcoal-filtered air.

13.4.8.18 Prunus persica (peach)

Badiani *et al* (1996) reported results of ozone exposure in OTCs in the summer of 1995. There was some discrepancy with results obtained in a comparable experiment in 1992, possibly due to lower levels of solar radiation in 1995. They calculated regression equations for various plant traits, such as firmness of fruit, number of fruitlets etc. Estimated AOT40s for 10% yield reduction ranged from 1.74 ppm-h to 9.41 ppm-h, with the economically most important one (for fruit firmness) at 2.96 ppm-h. This suggests that fruit trees may be particularly sensitive, but confirm the difficulty of setting critical levels for plants grown in varying ambient conditions, where more than one factor may be affecting growth response.

13.4.8.19 Agricultural crops (general)

Ashmore (1984) showed that legumes are the most ozone-sensitive economically important group. Brassicas, lettuce and beet are relatively resistant. Grasses and cereals are intermediate. He grouped agricultural crops by sensitivity (very, moderately, slightly sensitive, and resistant). Heck *et al*

Generally, only one cultivar of each was studied, but differences in response were apparent between years even in the same cultivar. Benton *et al* (1995) reported on experiments conducted in 17 European countries to determine which species develop visible injury or reduced biomass. In 1994, injury was noted in *Trifolium subterraneum, T. repens, Phaseolus vulgaris*, tomato, soybean, watermelon and tobacco. Yield reduction was checked by spraying plants with EDU in ambient ozone conditions. Biomass of subterranean clover shows foliar injury at high ambient ozone concentrations, but no yield reduction. Adams *et al* (1988) plotted ozone (ppm) 12-h averages against yield of soybeans, cotton, forage, corn and alfalfa and compared it with 7-h averages for spring and winter wheat, sorghum and rice.

13.4.9 Mediterranean Crops

Visible injury to plants and crop losses attributable to this pollutant have been described in various Mediterranean countries (Velissariou *et al*, 1996). Nevertheless, the effects of ozone on plants in the Mediterranean region are still poorly understood due to the scarcity of monitoring records in rural sites, and to the lack of knowledge on the sensitivity to ozone of native species and ecosystems. This is further complicated by the fact that the richness of species of this region is overwhelmingly high, a great diversity of species and life-cycle strategies exist.

Xerophytic vegetation, which may be considered the typical Mediterranean vegetation, shows a distinct behaviour in spring and summer (Gimeno *et al*, 1994). In the latter season, more resistance to gas exchange and therefore more resistance to ozone uptake can be forecast, so the actual dose of ozone would be much lower than the exposure, and less injury from ozone would be expected. However, other mechanisms might be involved as well since, in a study performed with Aleppo pine seedlings, ozone filtration during summer periods induced a greater increase in the carbon assimilation rate than in seedlings exposed to ambient levels (Elvira and Gimeno, 1996). Moreover, plant peroxidases increased in ozone fumigated plots during the summer. In the same experiment, ozone exposure induced a lower winter recovery of chlorophyll levels.

In Mediterranean areas, it is not easy to define a growing season, or a period in which the vegetation might show a higher sensitivity to ozone. For example, *Pinus halepensis* shows two flushes of growth per year in most of its natural locations. In an experiment performed in Greece, the Aleppo pine seedlings continued to grow even during the summer. Although many Mediterranean species blossom in spring time, there are many which do so at different times: in winter, in summer and autumn, in autumn, from autumn to spring, or even throughout the year.

The sensitivity of the different Mediterranean species was presented by Gimeno *et al*, (1994), although this list has since been modified. Grapevines have been studies by Soja (1997), showing sensitivity to ozone. Olives are considered to be tolerant (Badiani, personal communication). Peaches are considered to be sensitive (Badiani *et al*, 1996). So, for the most important Mediterranean species, their sensitivity class is as follows:

Citrus spp:	sensitive
Olives:	tolerant
Grapes:	sensitive

As for forest species, *Pinus halepensis* has been shown to be tolerant to ozone, although there are a number of indirect effects that have to be considered. These effects, which also apply to agricultural crops, include the increase of sensitivity to drought stress induced by ozone (Inclán *et al*, 1996; Gerant *et al*, 1996) or the increase in sensitivity to viruses (Porcuna, personal communication).

Until now, only effects on productivity have been considered. However, ozone may also affect product quality, or the ripening time of the crop (Bermejo *et al*, 1996). These effects may produce higher economic damages than productivity losses, particularly for high-value crops, such as oranges, tomatoes, or other horticultural crops typical of Mediterranean regions.

Table 13.5 Estimated percentage crop yield losses with ambient ozone and three ozone standard scenarios for 1984 (taken from Olszyk *et al*, 1988).

Crop	Standard scenarios			Actual ambient
	0.05 ppm	0.06 ppm	> 0.10 ppm = 0.10	
Alfalfa hay	7.8	8.8	8.6	8.9
Barley	0	0	0	0
Beans, dry	17.5	20.9	22.7	23.5
Corn, field	1.1	1.4	1.5	1.5
Corn, silage	1.5	2.2	2.6	2.9
Corn, sweet	5.4	6.1	5.9	6.1
Cotton	12.2	16.2	18.3	18.8
Grapes, all	15.9	20.8	21.9	22.4
Lemons	17.7	19.8	19.1	20.4
Lettuce	0	0	0	0
Onions, all	21.4	24.0	23.4	24.4
Orange	14.8	18.7	18.6	19.8
Rice	2.0	2.1	2.2	2.2
Sorghum, grain	0.4	0.5	0.6	0.7
Spinach	3.5	3.5	3.5	3.5
Strawberries	0	0	0	0
Sugar beets	0	0	0	0
Tomatoes, fresh	2.9	2.9	0	2.9
Tomatoes, proc	1.9	2.8	3.9	4.2
Wheat, all	0.8	0.8	0.8	0.8

13.4.10 Non-food Economic Crops

13.4.10.1 Nicotiana tabacum

Visible ozone injury is apparent if exposed to 50 ppb ozone for only 3 h (Menser and Heggestad 1966). Brewer *et al* (1988) predicted losses of between 3 and 11% in yield under ambient ozone levels in the Tennessee Valley (see winter wheat above). Gimeno *et al* (1995) compared 12-h day ozone exposure with 7-h day. Yield was reduced by an additional 10% in the 12-h day compared with the 7-h day receiving proportional addition. Heagle *et al* (1987) plotted marketable leaf weight against seasonal 7 and 12-h per day ozone concentrations in the range 25 to 125 ppb. There was a steep linear decline from over 300g at 25 ppb (CF) to less than 200g at 125 ppb.

13.4.10.2 Sunflower

Heck *et al* (1986) exposed seedlings from 14 days of age to 100 or 200 ppb for 12 days. The two concentrations resulted in 11% and 32% decrease in plant dry weight respectively. Roots were affected more than shoots.

13.4.11 Native Vegetation

13.4.11.1 Plantago major

The total weight of plants from 3 different populations was reduced to a different extent by different ozone treatments. The greatest reduction was in plants grown in 35 ppb and moved to 70 ppb for 3 days per week. Compared with CF grown plants, the reduction was 29, 27 and 16 % in the three different populations (Pearson *et al*, 1996).

13.4.11.2 Mixed herbaceous species.

Reiling and Davison (1992) exposed 32 taxa to 70 ppb ozone for 2 weeks at 7 h per day. Fourteen species showed a significant reduction in relative growth rate (RGR), but there was a wide range of response from no change to 24% for *Plantago major*. Significant effects were shown for both monocots and dicots and there were marked differences between genera e.g. there was nearly as much variation between different *Plantago* and *Rumex* species as there was across the whole group. Ashmore (1984) looked at the relative response of different native families. *Papilionaceae* was the most sensitive, *Chenopodiaceae* and *Umbelliferae* were also relatively sensitive, while *Graminae* and *Cruciferae* were intermediate. *Compositae* species were the most resistant. Of ecological types, species of cultivated land and calcareous habitats were relatively sensitive and species of acid, nutrient -poor habitats tended to be relatively resistant. *Molinia*, *Nardus*, *Sieglingia* and *Deschampsia flexuosa* were resistant, *Agrostis gigantea*, *A. stolonifera*, *Avena fatua*, *Phleum pratense* and *Catapodium rigidum* were highly sensitive. Ashmore *et al* (1995) also studied the response of calcareous grassland species to four treatments: CF, CF plus 50, 70 and 90 ppb ozone. Artificial mixtures were set up with 2 different grass species also in monocultures. There was no significant effect of AOT40 on biomass, although species composition altered in the *Festuca ovina* mixture, but not in the *F. rubra* one. Within the forb component, only *Leontodon* *hispidus* biomass declined with AOT40. Short-term experiments with species mixtures were screened by Ashmore *et al* (1996). They were conducted in a set of 8 closed chambers in a glasshouse. Ozone at 80 ppb was added for 8 h per day to half the chambers. Total ozone dose ranged from 3.96 to 4.54 ppm-h in the different experiments. Eight to ten week-old seedlings of 21 grasses and 17 forbs were screened. Percentage change in above-ground biomass ranged from a reduction of 29.1% in *Galium saxatile* to an increase of 40.6% in the grass, *Koeleria macrantha*. In general, forbs were more sensitive than grasses, indicating that ozone may be important in modifying grassland species composition. While only one of the grasses, *Holcus lanatus*, showed a significant growth reduction, 7 of the forbs did so. Bergmann *et al* (1995) studied the growth responses of native herbaceous species to ozone exposures in four treatments: CF, CF plus 30, CF plus 70 ppb per 8-h and CF plus 60% of O_3 concentration in ambient air for 24 h plus 30 ppb for 8 h. *Malva sylvestris* was the most sensitive species, also *Cirsium vulgare*. The six species of *Compositae* tested were all ozone-sensitive, in contrast to the data of Ashmore *et al* (1988).

13.4.11.3 Mixed herbaceous and dwarf shrub species.

Pleijel *et al* (1995) looked at a threshold level of 40 ppb above AOT40 for crops and forest trees. For natural vegetation, there are relatively few data, but some fast-growing, herbaceous species appear to be highly sensitive. Sensitivity to ozone may depend on how much of the maximum potential growth rate is realised; this is influenced by genetic constraints which are strong in stress-tolerators and artificially small in bred plants, but also by the environment in which the plants grow. Lab-grown, bred plants would then be the most sensitive, and stress-tolerators, growing in low productive environments, the least sensitive. Results of twenty-seven Swedish native herbs, which were exposed to CF, NF and NF+ (1.5 x filtered air), showed that the response to ozone was very small, although there was a weak trend towards slower growth with higher ozone concentrations. Eighteen of 27 species had a higher net growth in CF than in NF+, and *Festuca ovina* showed growth stimulation. There was visible injury in 3 species (*Dactylis glomerata, D. aschersoniana* and *Phleum alpinum*). Tables from Mortensen (1992) showed percent dry weight reduction at 55 ppb ozone for 7 h per day for 5 weeks compared with 10 ppb. Seven grasses showed a range from NS in *Agrostis tenuis* and *Lolium perenne* to 45% reduction in *Phleum pratense*. Data from Mortensen (1993) showed ozone effects at 86-96 ppb on growth of 19 subalpine plant species. Five species showed no effect, e.g. *Cirsium palustre* and *Salix herbacea*; 3 spp including *Rumex acetosa* showed an increase in growth (but only at 40-53 ppb) and 13 spp showed a decrease, ranging from 14% in *Leontodon autumnalis* and *Oxyria digyna* to 99% in *Phleum commutatum*, which was also sensitive at 40-53 ppb. Few of them would be adversely affected by current summer concentrations of ambient ozone in South Norway, but the particular sensitivity of *Phleum commutatum* suggests its suitability as a bioindicator. Data from Mortensen (1994) showed no effect of 78 ppb 8h per day for 40-56 days in *Eriophorum angustifolium*, *Festuca pratensis*, *Geranium sylvaticum*, *Vaccinium myrtillus*, *Juniperus communis* and *Agrostis tenuis*. *Trifolium pratense* and *Carex atrofusca* were affected at 43 ppb. Mortensen and Nilsen (1992) showed a decrease in growth of *Plantago lanceolata* at 50 ppb 8h day^{-1}. *Polygonum viviparum* and *Silene acaulis* responded at 80 ppb, but a number of other species showed no effect e.g. *Chrysanthemum leucanthemum, Solidago virgaurea, Betula nana, B. pubescens, Calluna vulgaris, Campanula rotundifolia* etc.

13.4.11.4 Calluna vulgaris

Foot *et al* (1996) studied the effects of ozone fumigation on the growth, physiology and frost sensitivity of *Calluna*. Plants were fumigated in OTCs, ambient CF air or 70 ppb O_3 for 8h per day and 5 days a week. Ozone exposure in winter increased frost sensitivity and the root:shoot ratio was reduced. However, there was no effect of ozone treatment in summer.

13.4.11.5 Sphagnum species

Potter *et al* (1996) exposed *S. cuspidatum*, *S. capillifolium, S. papillosum* and *S. recurvum* to acute ozone episodes, i.e. 150 ppb for 6 hrs at low temperatures. Although 3 species were not sensitive, the fumigation caused a significant reduction in photosynthesis in *S. recurvum*, as measured by carbon dioxide assimilation and chlorophyll fluorescence. It is likely that bryophytes, by the nature of their morphology and physiology, will be susceptible to short-term acute ozone episodes.

13.4.12 Ranking Plant Response to Ozone

In this report we have attempted to extract the most recently refined estimates of the impacts of ozone on major vegetation types and agricultural crops, and to indicate where data are still only very sketchy, or lacking. It is possible to make some preliminary rankings of differential sensitivity of different types of vegetation to ozone. However, there are a number of factors that need to be taken into account even in making these broad generalisations, such factors were discussed in section 13.4.2

The major economic output for agriculture is livestock production (milk, cattle and pig production are the three most important products, accounting for over 40% of agricultural production in the European Community in 1989). Beef production will be indirectly affected by the effects of ozone on pasture species, particularly *Lolium*. Sheep production (<3 % of the total) will be affected similarly by the impacts, if any, on pasturage, but also by ozone effects on upland grass species. For pigs and poultry (together > 16%), impacts would require a global scenario, as foodstuffs are imported from all over the world. None of these impacts, however, include any direct effects on animals themselves.

Except where more accurate data are available, the critical doses have been set somewhat arbitrarily as follows:

- For sensitive plants, at the level determined for wheat
- For very sensitive plants, at half that value
- For slightly sensitive plants, at twice the level
- For tolerant plants, ozone impacts can be ignored

13.4.13 Recommendations Based on Broad Vegetation Categories

In all the examples given in this report, critical doses are those calculated for a 10% loss in yield, unless otherwise stated.

13.4.14 Broad-leaved Trees

13.4.14.1 Fagus sylvatica

A critical dose of 10 ppm-h over 40 ppb, accumulated in the daylight hours of a 6 month growing season, has been agreed (Kärenlampi and Skärby, 1996). In the view of Kärenlampi and Skärby (1996) this should not be used to estimate quantitative biomass loss or to evaluate economic effects of ozone on forests, but may be used for modelling and mapping exceedence. In the context of the ExternE Project this distinction looks rather arbitrary, and creates a danger that policy makers will not be provided with data in as transparent a form as they might wish. The Project team do not believe that there is a problem with quantification so long as uncertainties are reported.

13.4.14.2 Fraxinus excelsior

Insufficient data, but the few studies done suggest that it is tolerant of ozone.

13.4.14.3 Quercus robur

Other data on conifers have been derived for American species. Kold *et al* (1997) found that *Picea rubens*, *Pinus ponerera* and *Sequiodendron giganeum* were less sensitive as adult trees than as seedlings.

Coniferous trees in general seem to be less sensitive than broad-leaves. However, the observation is based on similar exposure patterns, and may not hold true if allowance is made for year-round needle exposure. Also, comparisons of broad-leaves and conifers based on American species include a number of experimental studies on *Populus* species, some of which have been identified as being particularly sensitive, and would therefore distort any overall pattern.

13.4.16 Agricultural Cereal Crops

13.4.16.1 Wheat

The most-studied of all plant species, it appears to be relatively sensitive (based on experiments with spring wheat). Use the dose response relationship discussed by Fuhrer (1996). There is a linear decline between relative yield of wheat and the AOT40 so that an AOT40 of approximately 3 ppm-h results in a yield loss of 5%, and 5.7 ppm-h in a 10% loss. However, this response is only appropriate when soil moisture is not limiting, and it should not be converted into a yield loss estimate, but only used to indicate the degree of risk. The equation is:

$$\text{Yield} = 99.7 - 1.7 \times O_3\ (\text{ppm-h}); \qquad r2 = 0.89$$

13.4.16.2 Maize

Appears to be tolerant of ozone. Ignore potential impacts.

13.4.16.3 Barley

Conflicting evidence suggests that barley, like maize , is an ozone-tolerant species. Alternatively an equation given by Fuhrer (1996):

$$\text{Yield} = 99.7 - 1.5 \times O_3\ (\text{ppm-h}); \qquad r2 = 0.97$$

suggests that sensitivity is similar to wheat. Test both scenarios.

13.4.16.4 Sorghum

Relatively tolerant, but set critical dose at twice that for wheat: 10 ppm-h.

13.4.16.5 Oats

Relatively tolerant, but set critical dose at twice that for wheat: 10 ppm-h.

13.4.17 Pasture Grasses and Clover

13.4.17.1 Mixed pasture species

Ambient ozone levels are sufficient to affect the species composition of semi-natural grasslands, and cutting or grazing may also influence it. Generally, if clover species are present, higher levels of ozone tend to reduce the percentage of the legume relative to the grasses, which may result in higher grass production.

Pasture grasses are not as sensitive as wheat, possibly because the pasture is cropped before the onset of natural senescence, which tends to be enhanced by ozone exposure (Pleijel *et al*, 1996). Critical doses have not been determined, but perhaps set them at a high level (20 ppm-h) for this exercise.

13.4.17.2 Clover species

There are differences between species; *Trifolium*

13.4.18.4 Leaf vegetables

These can generally be grouped as relatively ozone tolerant. Ignore potential impacts.

13.4.18.5 Fruit trees and grape vines

NCLAN studies all indicated that they are more sensitive than wheat, although a European study of peach trees illustrated the difficulty of setting critical doses for plants in varying ambient conditions, where more than one factor may be affecting the growth response. Use critical dose for wheat.

13.4.18.6 Soft fruit

Strawberries appear to be tolerant; watermelon showed some visual damage, which was enhanced by sulphur dioxide. Use critical level of twice that for wheat, i.e. 10 ppm-h. Economically important agricultural crops not covered in this report, or for which only broad generalisations can be made, include rice, sugar beet and other root crops, and a number of Mediterranean crops such as peppers. However, carrots and peppers have both shown significantly reduced yields in NCLAN experiments.

13.4.19 Non-food Agricultural Crops

13.4.19.1 Tobacco

All studies suggest that this is a sensitive species. Ozone damage is shown by yield losses and by leaf damage. Set interim critical dose at half that for wheat (2.9 ppm-h).

13.4.19.2 Sunflower

NCLAN studies showed that seedlings responded by reduced growth, although roots were more affected than shoots. Use critical dose for wheat.

13.4.19.3 Agricultural crops in general

Table 13.21 shows the Weibull coefficient derived for the non-linear responses of various agricultural crops to daily ozone dose in various experiments during the NCLAN study.

13.4.20 Native Vegetation

The studies that have been done all suggest wide variation between different families, different genera and species, but also at the sub-specific level. There is also evidence of rapid genetic adaptation in some genera where detailed studies have been carried out over more than one generation, e.g. Plantago spp. While it would be inappropriate to attempt rankings, there are some species that appear to be particularly sensitive, which it has been suggested could be used as bio-indicators (e.g. *Phleum commutatum, P. pratense, Holcus lanatus*). In general, species of cultivated land and calcareous habitats are relatively sensitive and species of acid, nutrient-poor habitats relatively resistant. This may have implications for livestock production on semi-natural pastures. Many members of the Leguminosae are sensitive, which may have also implications for long-term soil fertility.

13.4.21 Preliminary Rankings

Table 13.6 is divided into 4 groups:

- Tolerant... set no critical dose;
- slightly sensitive...set dose at 10 ppm-h;
- sensitive...set dose at 5.7 ppm-h;
- ultra-sensitive...set dose at 2.9 ppm-h.

Species	Tolerant	Slightly Sensitive	Sensitive	Very Sensitive
Trees:				
Fagus sylvatica		V		
Fraxinus excelsior	V			
Quercus robur		V		
Betula spp		VT		
Populus spp				
Eucalyptus globulus				
Other eucalyptus				
Picea abies	V			
Pinus spp	V			
	Broad-leaves tend to be more sensitive than conifers in general			
Agricultural crops:				
Wheat			V	
Barley	V	....	V	
Sorghum		V		
Maize	V			
Pasture grasses		V		
Clover			V	V
Potato			V	
Soybean		V		
Beans			V	
Tomato			V..........................	V
Leaf crops	V			
Fruit trees (also citrus)			V	
Soft fruit	V			
Grapes			V	
Olives	V			
Rice		V		
Non-food agricultural crops:				
Tobacco				V
Sunflower			V	

Species	Tolerant	Slightly Sensitive	Sensitive	Very Sensitive
Native vegetation (studied species):				
Molinia caerulea	V			
Nardus stricta	V			
Deschampsia flexuosa	V			
Agrostis gigantea				V
A. stolonifera				V
A. tenuis		V		
Avena fatua		V		
Phleum pratense				V
P. commutatum				V
Holcus lanatus				V
Festuca ovina	V			
F. pratensis	V			
Dactylis glomerata			V	
Koeleria macrantha	V			
Eriophorum angustifolium	V			
Calluna vulgaris		V		
Betula nana	V			
B. pubescens	V			
Vaccinium myrtillus	V			
Juniperus communis	V			
Malva sylvestris				V
Leontodon hispidus		V		
Rumex acetosa	V			
Galium saxatile				V
Trifolium pratense			V	
Plantago lanceolata			V	
P. major		V	V	

Different results have been obtained from different studies, due largely to the fact that screening experiments are carried out under a particular set of conditions, but the family Leguminosae seems, in general, to be sensitive, and Graminaceous species are generally more tolerant than forbs.

13.5 Sulphur Dioxide

13.5.1 Background

The critical level for sulphur for agricultural crops has been set at 30 µg m^{-3} for both annual and winter means, as vegetation is more sensitive to SO_2 in the winter months. Forests and natural vegetation are more sensitive, so the level for them has been set at 20 µg m^{-3} (Jäger *et al*, 1992). Although sulphur is often referred to as the fourth major plant nutrient for agricultural crops (Syers *et al* 1987), more concern has centred on the harmful effects of acid deposition on natural ecosystems and acidification of waters. Historically, anthropogenic emissions of sulphur dioxide increased in Europe to a peak in the early 1970s, and since then have been declining gradually (McGrath and Zhao, 1995). The UK Acid Precipitation Monitoring Network has monitored sulphur dioxide in air at up to 30 locations in the UK.

Total deposition of sulphur in 1992 over much of the country was less than 30 kg ha^{-1} (McGrath *et al*, 1996). Depositions of more than 40 kg ha^{-1}, which were common, now probably only occur close to coal-burning power stations without flue desulphurisation equipment. Most rural areas probably receive between 5 and 15 kg ha^{-1} y^{-1} (Campbell and Smith, 1996).

Optimum rates of fertiliser S for oilseed rape are 20-30 kg ha^{-1}, and 10-20 kg ha^{-1} for cereals

13.5.4 Agricultural Crops

13.5.4.1 Wheat

McLeod *et al* (1991) used an open-air fumigation system to study the effect of SO_2 on winter wheat. There was a trend for decreased crop dry wt, leaf area and tiller density in the SO_2-fumigated plots during the winter and spring, although it had disappeared by the final harvest. At lower concentrations, there were yield increases, presumably due to SO_2 providing sulphur as a nutrient. Kohut *et al* (1987) found no effect of SO_2 by itself, or interactively with linear or non-linear components of the ozone dose-response function in winter wheat.

13.5.4.2 Maize

Miller (1988) demonstrated that SO_2 affected the photosynthetic rate in a range of crop plants. Compared with a zero control, SO_2 at 1 ppm (1000 ppb) caused an 80% reduction in photosynthetic rate of maize. This is a very high application level, however. Ashenden *et al* (1996) screened a range of plants for sensitivity to SO_2 at 100 ppb for 83 days exposure. Under those conditions, maize was not sensitive.

13.5.4.3 Barley

Winter barley was exposed to controlled releases of SO_2, ranging from 40 to 200 ppb above ambient (Baker *et al*, 1986). In winter and early spring, growth of the plants was depressed at high SO_2, but later recovered; in spring and summer, low concentrations stimulated growth above that of the controls, but there was no grain yield increase. Figure 4 of the paper shows linear regressions based on % reduction in grain yield as a result of SO_2 exposure from crop emergence to maturity, with SO_2 measured as mean annual concentration, ppb:
% reduction in grain yield = + 9.349 -0.69 SO_2 (6.9% loss per 10 ppb, equivalent to 0.64 t ha, and as a result of exposure to the average concentration during fumigation:
% reduction in grain yield = + 0.906 -0.215 SO_2 (2.2% loss per 10 ppb, equivalent to 0.19 t ha^{-1}).

Adaros *et al* (1991) summarised the single and interactive effects of ozone, NO_2 and SO_2 on two spring sown varieties of wheat and barley in OTCs. Of the main treatments, ozone caused the most growth reduction, particularly in wheat. Only one of the barley varieties was affected. Significant interactive effects were observed in length of ears, 1000-grain weight and weight of straw. Interactive effects were mostly antagonistic. Synergistic effects were however seen in wheat: the detrimental effect of ozone was amplified by either SO_2 or partially NO_2, or both together. Ozone -derived yield losses may be enhanced by low concentrations of other pollutants. Thus results from OTCs should consider interactive effects of mixtures present in ambient air.

McLeod *et al* (1991) demonstrated a decreased yield of winter barley in an open air fumigation system. There was a trend for decreased crop weight, leaf area and tiller density in winter and spring at the highest treatment applied, which exceeded the current range of annual mean SO_2 levels for urban sites. There was a yield increase at lower concentrations, presumably due to SO_2 providing sulphur as a nutrient.

Weigel *et al* (1990) concluded that SO_2 concentrations in the range of 50-90 µg m^{-3} (19-35 ppb) are potentially phytotoxic to some crop species. Two summer barley cultivars showed impaired yield at SO_2 concentrations > 100 µg m^{-3} (38 ppb); yield was reduced by 30-52% in exposure periods ranging from 49 to 96 days.

Data taken from this paper were subjected to linear regression analysis (European Commission, 1995):

% yield loss = + 10.92 - 0.89 SO_2 (SO_2 as ppb, calculated for 10 data points).

Ashenden *et al* (1996) showed a 37% reduction in total leaf area when exposed to 100 ppb SO_2 for 83 days.

13.5.4.4 Sorghum

Miller (1988) demonstrated how SO_2 affected the photosynthetic rate of sorghum. Compared with a zero control, the rate at 1000 ppb declined by about 20%. However, the treatment level was unrealistically high.

13.5.5 Pasture Grasses

13.5.5.1 Lolium perenne

Ashenden *et al* (1996) exposed *Lolium perenne* and *Agrostis capillaris* to combinations of gaseous pollutants and acid mists for 18 and 22 weeks respectively. Exposure levels of mixtures with SO_2 were:

40 ppb SO_2 + 40 ppb NO_2

40 ppb SO_2 + 40 ppb NO_2 + 40 ppb O_3 (with additional short peaks at 80 and 110 ppb).

Response to exposure levels was compared, with all gaseous pollutant mixtures causing a substantial reduction in *Lolium* dry wt, but it was less than additive in SO_2 + NO_2 + ozone. *Agrostis* was more resistant.

Roberts (1984) analysed dose response relations from the long-term effects of sulphur dioxide on 21 crops, but many data sets related to *Lolium perenne*. The most appropriate functional form was a linear dose regression based on studies that used constant concentrations e.g. SO_2 alone, in chambers. See Table 1 of his paper for a summary of results of regression analyses with data from various experiments, and European Commission (1995: pp 238-239).

Differences between species indicated that *Dactylis glomerata* and *Poa pratensis* were sensitive, as was one cultivar of *Phleum pratense*. *Lolium multiflorum* consistently appeared to be resistant, which contrasts with its response to ozone. L. perenne, which accounts for over 50% of new agricultural swards, is intermediate in sensitivity. Dose responses to exposure to ppb SO_2, calculated for that species, were:

% yield loss = + 2.75 - 0.18 SO_2.... for 45 data points, based on results for chamber exposures of more than 20 days at concentrations between 15 and 200 ppb SO_2.

% yield loss = + 7.33 -0.21 SO_2.... for 33 data points for chamber exposure of more than 20 days at concentrations between 15 and 200 ppb with more than one air change per minute.

This compares with an analysis of data merged from results for 9 species:
% yield loss = + 6.40 -0.31 SO_2.... for 35 data points for chamber exposure of more than 20 days at concentrations between 17 and 200 ppb, or an analysis of data from 21 species:
% yield loss = -3.18 -0.188 SO_2....for 115 data points for chamber exposure of more than 20 days at concentrations between 15 and 200 ppb.

Variation in response was caused by differences in pollutant flux, growth rate of the exposed species (a greater effect at low irradiance and short days), plant age (seedlings are more sensitive in some species, less in others), plant density, and interactions with other pollutant gases. A problem with the derived equations is that they do not extend to low SO_2 concentrations (between 15 and 35 ppb, depending on the equation) which correspond to a zero yield reduction. As already discussed in section 1.1, there may be growth stimulation below these concentrations, which would not be accurately reflected in the derivation of linear relationships.

Bell *et al* (1979) showed that concentrations as low as 15 ppb SO_2 could depress yields without visible symptoms in *Lolium perenne*.

SO_2 and NO_2 have been studied in combination, and percentage reductions in growth of *Lolium multiflorum* and *Phleum pratense*, when the plants were fumigated over winter and harvested in early spring, usually showed evidence of a synergistic effect, which was not apparent during summer, so there may be some interaction with cold stress (Ashenden and Williams, 1980). Concentrations were 68 ppb NO_2, 68 ppb SO_2 and 68 ppb of both. *Poa pratensis* growth showed a marked curvilinear dose response to mixtures of SO_2 and NO_2. Between 1.5 and 7.5 ppm days, dry weight as a percentage of the control declined from over 90% to about 20%.

Ashenden (1978a, b) exposed *Dactylis glomerata* to 110 ppb SO_2 for 4 weeks in a wind tunnel fumigation. The pollutant caused significant reductions in numbers of tillers and green leaves, leaf area, root/shoot ratio and dry weight. The % reductions were greater than those for *Lolium perenne*.

13.5.6 Agricultural (Fruit and Vegetable) Crops

13.5.6.1 Soybean (Glycine maxima)

Miller (1988) noted that 1000 ppb SO_2 caused an 80% reduction in dry weight of soybean, compared with a zero control (NB the high level of SO_2 for the treatment). Heagle and Johnson (1979) found a range of synergistic, additive and antagonistic effects of ozone and SO_2 on soybean. Synergistic effects were usually shown when individual doses were too low to have effect, but antagonism occurred when individual effects were severe.

13.5.6.2 Beans

Vicia faba appears to be particularly sensitive. Black and Unsworth (1979) measured a 7% reduction in yield when exposed to 35 ppb SO_2 for only 7 h. Two bean (*Phaseolus vulgaris*) cultivars showed growth reductions of between 10 and 26% when exposed to SO_2 concentrations in the range 50-190 µg m^{-3} (Weigel *et al*, 1990). In European Commission (1995), a linear regression against ppb SO_2 was calculated from the data of this experiment:

% yield loss = -0.93 - 0.60 SO_2 ....for 10 data points

Ormrod *et al* (1981) and Black *et al* (1982) studied response of broad bean to ozone with and without SO_2 added at a concentration of 40 ppb for 4 h. Results suggest a synergistic response to concentrations of ozone between approximately 60 and 150 ppb and an additive or antagonistic response above that level.

13.5.6.3 Rape

Weigel *et al* (1990) showed that dry matter and yield of rape was unaffected by exposure to sulphur dioxide for a growing season. Four SO_2 treatments of approx. 19, 33, 50 and 67 ppb were compared with CF air with about 3 ppb. The results corroborate those discussed in section 1.1, showing that oilseed rape has a high S requirement. Adaros *et al* (1991) studied interactive effects of ozone, SO_2 and NO_2 on growth and yield of spring rape. Medium levels of SO_2, of around 17 ppb, stimulated pod weight by up to 33%, while higher concentrations (35 ppb) caused a decline of over 12%. Most interactions were antagonistic, as some of the detrimental effects of O_3 were mitigated by these pollutants.

13.5.6.4 Water melon

Eason and Reinert (1996) measured the leaf area response of watermelon exposed to different sulphur dioxide levels with and without ozone. SO_2 intensified ozone suppression.

13.5.6.5 Cucumber and tobacco

Mejstrik (1980) fumigated cultivars of tobacco and cucumber for 4 weeks with 20 ppb SO_2. There were significant reductions in the fresh weights of green leaves, shoots and roots, root/shoot ratio and all dry wt fractions. Cucumber was more severely affected than tobacco.

13.5.6.6 Range of crop species

Fujiwara (1975) ranked species' susceptibility to SO_2 as:
tomato< lettuce< cabbage< soybean < pea< turnip < sweet potato< rice < buckwheat.

13.5.6.7 Sunflower

Reports of synergistic interactions of ozone and SO_2 on growth have prompted investigators to look at photosynthesis for a possible explanation. Furokawa and Totsuka (1979) observed a synergistic decrease in photosynthesis of sunflower at concentrations of 20 ppb each of SO_2 and ozone. However, synergistic reductions in photosynthesis have not yet been conclusively tied to synergistic growth reductions. Other data suggest that sunflower is a particularly sensitive species.

13.5.6.8 Mediterranean plants

There has been very little research on the response to SO_2 of plants which grow in Mediterranean climates (Wilson, 1995). Because of drought stress adaptations, which enable plants to survive in the hot, dry summers, it is likely that the vegetation's response to SO_2 may be different from vegetation growing in cooler, moister climates. It is possible that common mechanisms underlie resistance of both drought stress and SO_2 stress, and extrapolation from temperate species may be inappropriate.

13.5.6.9 UK Native species

Ashenden *et al* (1996) screened 41 herbaceous plants for sensitivity to 100 ppb SO_2 for 83 days, with some higher peaks. Total leaf area was reduced in 7 species in response to SO_2. Slow-growing, perennial species were less susceptible to SO

13.5.7.3 Agricultural (fruit and vegetable) crops

Vicia faba and other leguminous crops appear to be particularly sensitive to SO_2. Oil seed rape is very tolerant, and ambient concentrations are often too low for optimum crop yield.

13.5.7.4 Native vegetation

Limited screening suggests that slow-growing perennial plants are less susceptible to SO_2 than fast-growing perennials or annual species.

13.5.7.5 Interactions with pests and pathogens

SO_2 can exacerbate the effects of insect herbivores, by a disproportionate effect on their predators and parasites. There may also be changes to the plants' biochemistry that makes them more palatable. These effects would be difficult to quantify, and would vary depending on the distribution of the pests.

13.6 Nitrogen Oxides

Nitrogen oxides appear to be less significant in terms of their direct pollutant effect on vegetation. A critical level of NO_x for all vegetation types was agreed upon in Egham (1992) as an annual mean of 30 and a 4 hour mean of 95 ug m^{-3}. This level is based on the assumption that SO_2 and/or ozone are close to their critical levels (Sanders *et al* 1995). The effect of nitrogen oxides has been stated to be neutral, except in its role as a chemical precursor of ozone, or as a source of acidification (Semenov, 1992). The effect of it as a pollutant has only rarely been studied in isolation, e.g. Adaros *et al* (1991) studied its impact on the growth and yield of spring rape on its own and in combination with ozone and SO_2. NO_2 alone had a slightly positive effect, and it ameliorated some of the detrimental effects of ozone, except on leaf area, in which the joint impact was synergistic.

Saxe and Muralin (1989) studied the impact of NO_2 and NO on *Picea abies*. All the mixtures and concentrations studied caused a reduction in photosynthesis and transpiration within one hour, but with widely different effects on different populations. In addition, night transpiration and dark respiration were affected. Dueck *et al* (1988) also noted large differences in response between different populations of *Agrostis capillaris*, *Nardus stricta* and *Lolium perenne* obtained from area differing in the level of ambient pollution.

Sensitivity to NO_x may be scaled as:

Natural vegetation > forests > crops

Nitrogen is of course 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).

13.7 Interactive Effects of Ozone, Sulphur Dioxide and Nitrogen Oxides

Mansfield and McCune (1988) summarised interactive effects as follows:

- More than additive effects of SO_2 and NO_2 in short and long-term experiments occur not infrequently.
- Considerable seasonal variation occurs in response and the effects may be greater at low temperatures. Wolfenden *et al* (1991) also showed that mixtures of the two gases adversely affected the frost hardiness of *Picea rubens*.
- Assimilate translocation to roots is suppressed to a greater extent by SO_2 + NO_2 than by SO_2 alone, even though NO_2 has no individual effect.
- Damage to epidermal cells has been observed when leaves are exposed to SO_2 + NO_2, which may prevent stomatal closure and affect the plant's ability to survive drought. This would presumably have further implications for the effects of ozone exposure. Conversely Omasa (1990) observed that there was a greater degree of stomatal closure in sunflower when exposure to SO_2 + NO_2 + ozone was increased.

Bender and Weigel (1993) used factorial designs to research the interaction of ozone with SO_2 and/or NO_2 on yield responses of bean, wheat and barley. Although the ozone effects were dominant, they were altered by low levels of the other air pollutants. Additive effects were most frequent, but when interactions occurred, the mode of interaction was mostly antagonistic, i.e. the presence of the other air pollutant reduced the impact of ozone.

Murray *et al* (1992) compared the response of barley and clover to mixtures of SO_2 and NO_2, ranging from less than 5 to 544 ppb SO_2 and <5 or 170 ppb NO_2 for 4 hr day-1 over 108 days. NO_2 halved plant dry weight in clover, but under the same conditions, it increased plant dry weight in barley by increasing ear weight, and grain number and weight. When barley or clover were exposed to SO_2 alone, growth decreased as SO_2 increased. Mixtures of NO_2 and intermediate SO_2 concentrations (55 or 149 ppb) caused a substantial increase in barley grain production, but did not affect vegetative growth. The differences observed between the responses of barley and clover to the gas mixtures could be due to their differences in access to sources of nitrogen, as clover is a nitrogen fixer. They suggested that the response could be due to the effect of the gases on detoxification, allocation, transport, use and storage of SO_2 and NO_2 derivatives.

In studies on spring rape, Adaros *et al* (1991) showed that the interactive effects of SO_2 and NO_2, particularly with ozone, was antagonistic. Some of the detrimental effects of ozone were mitigated by the presence of the other air pollutants. However, while 56 ug m^{-3} SO_2 increased yield in isolation, it increased the yield loss caused by ozone from -16.8% to -21.4%. Leaf area was the only measured parameter that was negatively affected by all the pollutants, and their interactive effect was synergistic.

Fangmeier (1989) exposed the herb layer of a melick- beech forest to SO_2, SO_2 + NO_2 or SO_2 +NO_2 + ozone. At the population level, responses to the fumigants ranged from undetectable in some of the grasses to severe loss of leaf area in some of the geophytes, due to early and increased senescence. He found it difficult to interpret the range of biochemical responses, as these varied with microclimate and stage of plant development.

- NO_x + SO_2 mixtures may have a positive or negative effect on plant growth, depending on the species.
- Interactions may be antagonistic, and some of the detrimental effects of ozone mitigated by the other pollutants.
- More than additive effects of SO_2 and NO_x can occur.
- Interactive effects may be greater at low temperatures, possibly because of the role of the individual pollutants in reducing frost hardiness.

13.8 Assessment of Impacts of Acidification on Agricultural Soils

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. So far as agricultural systems are concerned, acidification from atmospheric inputs must be seen against this background of alternative acid inputs. 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. 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.

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);

$$H_2SO_4 + CaCO_3 \rightarrow CaSO_4 + H_2O + CO_2$$

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 $CaCO_3$ is sufficient to neutralise 2 kg H^+. Accordingly the total acidifying pollution input on soils that require lime was multiplied by 50 to give the amount of lime that was 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.

13.9 Interactions with Pests and Pathogens

There is strong evidence of changes in insect herbivory induced by air pollution. Most of the experiments performed have only looked at the response of the insects, and generally, it appears that there is more plant damage by insect herbivores in the presence of SO_2. Little is known about long-term effects of population dynamics, but it is possible that air pollution has a dual role in reducing parasitism and predation on the pests and increasing general suitability of plants to them (Riemer and Whittaker, 1989).

Field observations in polluted areas have indicated changes in abundance of pests and pathogens, and in some cases a causal link has been shown in controlled experiments. Experimental results with wheat, barley, broad beans, peas and brussel sprouts indicated a remarkably consistent increase in the growth rate of insect pests when the host plants were fumigated with 100 ppb SO_2 or with 100 ppb NO_2 (Houlden *et al*, 1990).

Bell *et al* (1993) reviewed impacts of SO_2, NO_2 and ozone on fungal pathogens and aphid pests. Tables 1 and 2 of the review summarise published data of the effects of ambient air compared with CF air on some plant species, and of some field observations on changes in insect pest abundance in relation to pollution levels respectively (see Table 13.7).

Gate *et al*, (1995) showed that the proportion of hosts parasitised and the searching efficiency of the parasitoid was affected by air pollution, particularly ozone. The implication is that the efficiency of the natural control enemy of many pest species is reduced.

Masters and McNeill (1996) studied the effect of NO_2 pollution on aphids of different varieties of beans. The susceptibility of the plant variety to insect herbivory could be significantly altered when it was subjected to pollutant stress.

Table 13.7 Field observations of changes in incidence of insect pests at polluted locations (taken from Bell *et al*, 1993).

To establish monetary values, an inventory was made of the cropped area per species for each province and the yield at ambient concentration levels. New prices were calculated for any changes in supply taking into account the price elasticity per species. Estimated yields were not simply valued using current market prices, but a model of supply and demand was used to obtain estimates in the changes in producers' and consumers' surpluses. Although these adjustments were made to allow for price effects, there were other effects that were ignored. The supply functions were not derived from an overall profit maximisation model of farmer response, so cropping patterns were not fully adjusted to allow for the differential environmental impacts of changes in pollutants on yields. Second, as the dose-response relationships are non-linear, this will affect the grossing-up of national estimates. Finally the lack of information on sensitivity of certain species in the Netherlands to the pollutants in question has probably lead to damage estimates that are too low.

AED (1991) found that the crop loss in the Andalusian region of Spain was in the order of 930 million Pesetas (ECU 8.7 million, 1987). This result was obtained using market prices only. Using market prices in conjunction with a demand and supply model, another estimate of 1.7 billion Pesetas (ECU 15.8 million, 1987) was found (including wheat and corn, all figures in 1987 prices), indicating that, as in the van der Eerden study, the consumer losses from price rises are a significant part of the costs of the air pollution. Although there have been several reservations about some of the functions used in this study, it represents the most comprehensive attempt at crop damage valuation in a European country. Since the 930 million Pesetas estimate is only 50%-60% of the estimate found by allowing for price changes, this illustrates the importance of calculating consumer and producer surpluses, in the case of the kinds of changes in yields that are being considered in that study.

13.10.1.2 Issues

The main issue of concern is the relevance of the price changes induced by the changes in production to this exercise.

13.10.1.3 Price changes and other adaptations

As demonstrated above, estimated damages will depend on how much prices are affected by the changes in concentrations. The earliest crop damage estimate studies simply multiplied the reduction in output attributable to the pollutant (according to the biological dose-response relationship) by the market price. This only gives a partial equilibrium analysis and can only be taken as a rough estimate. It ignores behavioural responses or price changes, which would be included in a general equilibrium analysis.

Indeed, in the context of higher pollution levels, producers are likely to switch to those crop varieties that can give a better resistance to the air pollutants in the region. Alternatively, the farmers may alter the use of other inputs such as calcium carbonate, which mitigates the pollution impact. In relation to price changes, a fall in the yield as a result of pollution will generally mean less output at a higher market price. Consumers will lose, when pollution increases, if the market prices are at all sensitive to output. This was shown clearly in the European studies referred to above. In addition to these studies, US studies have also shown the importance of the price effects, where the percentage of the total loss attributable to consumers ranged from 50% (Adams, Hamilton and McCarl, 1986) to 100% (Adams and

McCarl. 1985). Given that producers can sometimes gain from an increase in air pollution, not including the consumer impacts can underestimate total losses and misrepresent the distribution of welfare effects. Producers, on the other hand, will lose or gain depending on the price elasticities of their particular crops. As an approximation, if the percentage reduction in quantity is greater (less) than the percentage increase in price then the producer will lose (gain). Shortle, Dunn and Philips (1986) and Adams and McCarl (1985) found that substantial increases in ozone increased the economic rents of corn belt growers. An American study by Adams, Crocker and Thanavibulchai (1982) found that using the simple multiplication approach gave valuation estimates 20% higher than if price and behavioural changes were taken into account. The authors used mathematical programming (consistent within the framework of standard microeconomic theory concerning firm and market behaviour), which is able to account more completely for the adaptations economic agents make to mitigate or exploit the opportunities of environmental damage.

As far as ExternE is concerned the question of importance therefore is to what extent the prices will change as a result of the changes in concentrations of pollution? If one assumes that only the plant in question will operate at the lower (or higher) level of emissions, the impact on overall production will be negligible and prices will not be affected. If, on the other hand, *all* power plants in a country were to change emissions, the impact would be more significant (although not again as much as is assumed in national studies).

The next issue is whether one should take the change in value, net of costs of production, or the gross value. Ideally what one wants to measure is the change in the value of the rents derived from the production. In the case where there is a change in an exogenous factor such as the deposition of pollutants, it can be shown that the change in the net income after deducting the costs of inputs is given by the *gross value* of the change in production. This conclusion is dependent on the assumptions that:

a) the producers are price takers,
b) there is no change in the prices of the inputs or outputs and, related to that,
c) the changes in the quantities are marginal.

For many of the crops that are affected, the prices are not determined by market forces but are influenced by the Common Agricultural Policy. In such cases, it has been argued by many economists that the changes in the value of the output should be valued at international or border prices and not the actual prices that are prevailing in the market in question (see, for example, Squire and van der Tak, 1975). The view taken here is that it is appropriate to use border prices as long as the reason for the divergence between the domestic and international price is not the correction of some externality. For major crops produced in the EU there is no direct externality involved and it therefore seems appropriate to use the international price in valuing the changes.

13.10.1.4 Recommendations

From the review of the European studies, the following recommendations are made:

- As an initial estimate, it seems reasonable to assume that prices are unaffected by effects of fuel cycle emissions, especially where only outputs in the plant in question will change. In future work, however, the issue of more general equilibrium changes should be explored;
- valuations should be made on the basis of gross value using international or border prices;
- a wide range of crops can be affected. Adequate dose response functions do not appear to be available for all of them, especially the Mediterranean crops such as tobacco, olives etc. Further research is needed to establish these functions. However, farmers and local communities claim that air pollution does affect yields and in many cases some information is available on yield impacts. This may be used as a 'first approximation' to obtain damage estimates.

13.10.2 Forests

There are many European studies of interest with respect to forest damage which include the estimation of commercial timber loss due to air pollution and also the loss in the recreational value of forestry. The major studies include:

- Nilsson (1991)
- NME (1988)
- Ewers *et al* (1986)
- Linden and Oosterhuis (1988)
- Navrud (1990)
- Hoehn and Winther (1991)
- Kristrom (1988)
- Grayson *et al* (1975)
- Everett (1979)
- Willis and Benson (1989)

13.10.2.1 Issues

There are several data limitations in this impact category, similar to those for crops. A major problem is a lack of specificity of results with respect to the precise pollution climate that a forest may be subjected to. In addition there are many difficulties in translating physical impacts into monetary damages. A forest has many functions including those of providing timber, other non-timber products, ensuring a habitat for animal and plant species and acting as a natural buffer against environmental shocks (e.g., global warming). It is thus not unreasonable to suggest that a significant portion of the total value of a forest is from non-market values. Quantification of forest damage is further complicated by issues of transferability. It is unlikely that estimated valuations will be applicable or transferable to other locations since the blame and causation of forest damage tends to be very 'site specific'. All physical impacts and valuations will vary according to location. The categories of damages are commercial timber, recreation and non-use functions. In this section attention is concentrated on the first two, with non-use functions looked at in Section 13.11.3.

Changes in the yield of commercially produced timber are simply valued by combining information on pollution-induced physical changes in the resource with details of market prices.

In the case of recreational damage, transferability to other regions or countries is much more limited than for many other categories of damages. The marginal value of forests as a recreational asset may be very different according to local demand and supply conditions. For example, Drake (1987) estimated a positive willingness to pay in Sweden for the preservation of open agricultural land (open space) in what is a densely forested area; the marginal value of forests was therefore negative. Such a result would be very unlikely in say, the Netherlands, where forests are more scarce and their recreational values have been shown to be largely positive.

Almost all recreational damage valuations rely on the contingent valuation method, in which the marginal impact of pollution on recreation values is very rarely assessed. The valuations obtained are thus average values. Furthermore, there are good reasons to think that the answers given to such questions express a wider concern for the effects of air pollution, not just forest damage (i.e. they ‘embed’ other values). This makes it incorrect to attribute the damages of the air pollution to forests alone, and adding the ‘forest’ damages to other damages involves double counting.

13.10.2.2 Main Results

Unfortunately, almost all the studies lack data on the baseline levels of pollutants and the land area of forest surveyed. Consequently, for the vast majority of studies no attempt can be made to establish an external marginal valuation for forestry based on per tonne of pollutant. NME (1988) calculated losses over 25-30 years from present production levels in Norway on the assumption that current ozone and acid depositions continue. Average timber losses came to NOK 1 billion (1988) or ECU 140,000 per year starting 25 - 30 years from now. This study also used expert judgements on the rate of loss of tree growth, a practice that is more common in this area of damage.

Another Norwegian study is that by Hoen and Winther (1991) who established a total value (both use and non-use) of multiple-use forestry and preservation of virgin coniferous forests in Norway. They found a mean annual WTP per household (in different sub-samples totalling 1204 persons) of ECU 16 - 46 (for multiple-use forestry) and a median of ECU 6 - 12. For the preservation of virgin forests a mean value of ECU 26 - 36 was established with a median of ECU 12 - 15. Most of the estimated value was due to the recreational value of the forest. Results show that the WTP for multiple-use forestry is (significantly) dependent on the experimental design in different sub-samples. This is not the case for the preservation values and so the authors note that this may be because ‘multiple-use forestry’ is too unfamiliar and vague a concept to respondents.

Linden and Oosterhuis (1988) looked at the continued damage by acid rain to 2010 in the Netherlands. If no appropriate abatement policy is adopted they assumed that this will lead to 80% of forest becoming damaged and 90% of heath becoming grassy vegetation. Linden and Oosterhuis (1988) surveyed a sample confronted with the assumption that 80% of forests will be severely damaged and 90% of heather will be crowded out by grassy vegetation by 2010 in the absence of an appropriate abatement policy. Timber losses were estimated at DFL 13.1 million per year (ECU 5.62 million, 1988). This is the expected loss if no measures against acid deposition/emission are taken. Maintaining current conditions leads to a WTP per household per month of DFL 22.83 (ECU 10.2, 1990). Extrapolated to the Dutch population reaches DFL 11.45 billion (ECU 5.13 billion, 1990) per year (which is a very large damage figure). Significant variables included income (level and changes), the perception of the acid rain problem, number of visits, age, education and social class.

Ewers *et al* (1986) focused on the recreational value of a forest and examined the rate of forest loss in West Germany over the period 1984-2060 (using different air pollution reduction scenarios over time). The study established a WTP per visiting hour per person of DM 4.87 (ECU 2.55, 1993) but this cannot be associated with a measured pollutant deposition.

Navrud (1990) looked at the recreational value of mountainous forest. Respondents were confronted with the proposal that the forest could be subject to 3 different management practices; clear cutting, selection forestry and no cutting (preservation). However, this does not relate to reduced recreational facilities resulting from air pollution.

The Austrian Ministry for the Environment made a national evaluation of forest decline attributable to air pollution. They established a total estimated forest loss at AS 4.5 billion per year (ECU 0.22 billion, 1983). 85% of the forestry losses came from items other than commercial timber.

A Swedish study by Kristrom (1988) studied the benefits of preserving selected regions of virgin forests, i.e., preventing commercial forestry. This established a WTP (open) of SEK 1014 (ECU 155.0, 1990) and a WTP (closed) of SEK 1005 - 2741 (ECU 153.6 - 419.0, 1990). Thus for 3 million Swedish households this gave a total WTP of SEK 3 billion - 8.2 billion (ECU 458 million - 1.25 billion, 1990). The WTP question was posed in a 'closed' format and an 'open' format. The closed format involves a model that describes the probability that the suggested bid is lower or higher than the person's actual WTP given in the conventional open format.

There are three major UK studies that estimated the recreational value of national forests. The results are summarised in Table 13.8. They show a consumer surplus per hectare ranging from around ECU 52/ha. to ECU 154/ha. As none of these figures can be related to losses of recreation value resulting from increases in forest damage from air pollution, they are of no use in the context of this study.

Table 13.8 The recreational values of forests from three major UK studies.

Study	Valuation Type	Results	Comments
Grayson *et al* (1975)	Consumer Surplus estimated for all Forestry Commission Forests aged 25+ (1969-71)	Consumer Surplus per Hectare: £30.50 (£1987) (ECU 52.9) Consumer Surplus per visitor: £0.33 (ECU 0.57)	Per hectare calculation is derived from the consumer surplus per visitor * all forest visitors divided by total area of land of FC land aged 25+ years
Everett (1979)	Recreational value of forest in Gwydyr, UK 1975/76	Consumer Surplus per Hectare: £64.60 (£1987) (ECU 112.1) Consumer Surplus per visitor: £1.82 (ECU 3.1)	Different estimates of annual visitor numbers produce different estimates of consumer surplus per ha
Willis and Benson (1989)	Consumer surplus was estimated for forest recreation on six Forestry Commission sites	Consumer Surplus: (a) For all six sites = £1.90 (ECU 2.92 per visitor, £31.78 (ECU 48.8) and £100.51 per hectare [ECU 154.2](see comment) (b) For Dalby only = £1.82 (ECU 2.8) per visitor £52.44 (ECU 80.5) For Dalby different estimates of annual visitor numbers produce different estimates of consumer surplus per hectare	For the consumer surplus per hectare values: £31.78 represents consumer surplus per visitor * total visitors to the sites divided by the total area of the sites. £100.51 represents consumer surplus per visitor * all forest visitors divided by the total area of Commission land aged 27+ years

Note: All ECUs quoted for £1990 where ECU 140 = £100

The most adventurous assessment of air pollution effects on forests is that by Nilsson (1991), carried out at the International Institute For Applied Systems Analysis (IIASA, 1991). However, the forest decline module used there is not robust (European Commission, 1995) and has not been used here. A factor of [2.7*lost timber value] was used to account for non-timber damages. This value was estimated from a survey of existing valuation studies, but again is not considered robust. Since most of the other studies are related to recreational forest value they cannot be transferred to the fuel cycle valuation. This includes checking CVM results against other valuation techniques and checking for biases in responses to questionnaires.

13.10.3 Natural and Semi-natural Ecosystems

This impact category is perhaps one of the most difficult to value. Four European studies that bear on ecosystem valuation have been identified:

- Dahle, L. *et al*. (1987)
- Hervik, A. *et al* (1986)
- Travers Morgan Economics (1991)
- Johansson (1987)

13.10.3.1 Issues

Contingent valuation is the only valuation technique for eliciting preferences for environmental assets that have no related market. Valuing biological diversity is a case in point. As mentioned in the previous chapter, there is a problem regarding the validation of an experimentally-created market. Some techniques are available but there is no obvious way to validate the estimated willingness to pay for conservation. One advantage of contingent valuation is that it can capture 'existence' and 'option' values whereas other valuation techniques tend to focus on use values. Table 13.9 below, adapted from Pearce *et al* (1992) shows the results of CVM surveys of studies valuing endangered or rare species. The results are interesting because of their broad consistency. Most of the valuations tend to cluster around ECU 7 (1990 prices) if we exclude the relatively high value for humpback whales. The range is ECU 1.4-15.4 excluding humpback whales and ECU 1.4-42.0 including humpback whales. The results suggest that:

- they do not represent large proportions of respondent income, and
- habitat appears to be more highly valued than species, which is to be expected since a wider array of benefits is being secured through conservation of habitat than through targeting individual species.

The transferability of per capita values of this kind is very limited. The values in one country are very much a function of local factors and it is inappropriate to take them out of context. However, the conclusion that *per capita* values for different species and different locations are not generally transferable remains.

Table 13.9 Per capita preference valuations for endangered species and prized habitats.

		$ Value	ECU Value
SPECIES			
Norway	Brown bear, wolf and wolverine	15.0	12.3
USA	Bald Eagle	12.4	10.2
	Emerald Shiner	4.5	3.6
	Grizzly Bear	18.5	15.3
	Bighorn Sheep	8.6	7.0
	Whooping Crane	1.2	1.0
	Blue Whale	7.5	6.2
	Bottlenose Dolphin	5.4	4.5
	California Sea Otter	6.0	4.9
	Northern Elephant Seal	8.1	6.7
	Humpback Whales	40 - 48	32.9 - 39.5
		49 - 64	40.4 - 52.7
HABITAT			
USA	Grand Canyon (Visibility)	27.0	22.3
	Colorado Wilderness	2.7 - 6.0	2.2 - 4.9
Australia	Nadgee Nature Reserve (NSW)	28.1	23.1
	Kakadu Conservation	40.0	32.9
	Zone, NT	28.6	23.5
UK	Nature Reserves	40.0	32.9
	Flow Country	28.6	23.5
Norway	Conservation of Riveres against hydroelectric development	59.0 - 107.0	48.6 - 88.1

Source: Pearce *et al* (1992), converted to ECU at £1 = 1.4 ECU
($ 1990 per annum per person, ECU 1990 per annum per person)

13.10.3.2 Main Results

The Norwegian study by Dahle *et al* (1987) estimated a WTP for the protection of the endangered brown bear, wolf and wolverine. An average WTP for all three creatures of US$ 15 (ECU 13.5, 1990 prices) per annum per person was found. The study by Hervik *et al* (1986) interviewed people regarding the conservation of rivers against the development of hydroelectric power plants. They found a WTP of US$ 59-107 (ECU 53.2 - 96.5, 1990) per person per annum. This latter result could be of use in valuing the loss of habitat and ecosystem function from hydropower development. Although that is not currently being looked at in the fuel cycle study, it will be part of the program in the future.

The most problematic values to elicit tend to be the non-user values, some of which are referred to as existence values. The CVM technique, which can work well for situations where a market environment can be simulated, do not work well in situations where it cannot. Thus non-use values must remain extremely uncertain even when used in the context in which they were carried out. Outside that context they have to be considered unacceptable at the present time.

Some attempts to value non-use in a specific context exist. For example, a UK study, using CVM methods, carried out by Travers Morgan (1991), estimated non-user values for wildlife and habitat in the Mersey Estuary. The construction of a barrage to capture tidal energy in the estuary would have adversely affected surrounding wildlife. Some non-user values were estimated by questioning people in Bristol and Sheffield, which were considered to be far enough away not to represent use values. A WTP was elicited from a sample of 300, which, if extrapolated across the UK, would give a value of ECU 146 million. The extension of the results across the UK must be problematic and the embedding problem (i.e. that people are expressing a general value for loss of wildlife) remains. Finally, as noted earlier, such a study is not transferable outside this context.

There are several studies that attempt to estimate the benefits of species and ecosystem functions but they have no bearing on valuing these losses in the context of energy production. (see Johansson, 1987). Others could be relevant but would need further data if they were to be used.

13.10.3.3 Recommendations

There are real problems with using existing studies to value biodiversity losses as a result of different fuel cycles. Results are of extremely limited transferability. Specific studies can be carried out, but they will not generally be able to provide credible estimates of non-use values. User value losses can be valued for each site, but they need specific studies, or they need values of losses taken from sites with very similar characteristics. These will be most relevant in the hydropower fuel cycle. Outside of those cycles, however, there is no data that can be used to value any loss of biodiversity, which, in any event would be indirect and difficult if not impossible to link to the fuel cycle itself.

13.11 Conclusions

1. Several of the major air pollutants are known to be capable of affecting plant growth, the pollutants of greatest concern being SO_2, NO_x, NH_3, O_3 and acidity.

2. Associated impact pathways are complex. A large number of variables, above and below ground will be affected. Knowledge of a number of effects is limited.

3. Direct impacts of acidic deposition on foliage may be limited. So far as forests and natural terrestrial ecosystems are concerned it seems likely that the most severe effects result from soil acidification and nutrification. Acidification can lead to depletion of nutrient base cations from the ecosystem and the accumulation of potentially toxic species, particularly Al. Nutrification can change the competitive balance in ecosystems, which has frequently evolved over greatly extended time scales

4. 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.

5. Mechanistic models are not available for assessment of threats to ecosystems at the scale required for this study.

6. Sensitivity of plants to acute pollutant doses is a poor guide to their reaction to chronic exposure.

7. 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 SO_2 and O_3 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:

- Interactions with pests;
- Interactions with pathogens;
- Interactions with climate;
- Effects of, and interactions with, other pollutants;
- Yield losses on crops other than those specifically covered by exposure-response functions.

8. 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.

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. Valuation of changes to natural ecosystems is not currently possible in this context. Alternative approaches based around the discussion of sustainability presented in Chapter 2 may be more appropriate.

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APPENDIX 13.1: NOTES ON DIFFERENT WAYS TO MEASURE OZONE EXPOSURE

One of the difficulties in interpreting and comparing vegetation response to ozone exposure is the wide range of different methods used to assess the ozone dose. Researchers in Europe have agreed to standardise using the dose calculated as AOT40, as described in the Introduction of this report. American studies, particularly those made during the NCLAN programme, have used other methods. The following notes summarise some of the methods used.

Lefohn (1992) discussed ozone standards and relevance for protecting vegetation. The SUM06 index is calculated by summing hourly average concentrations across a fixed 3-month period. Tingey *et al* (1991) reported that a SUM06 value of 24.4 ppm-h was estimated to cause a 10% yield loss in half the cases they investigated, but it is difficult to relate this dose to AOT40. SUM-06 is the accumulation of daily 1-h average ozone concentrations equal to or greater than 60 ppb throughout the growing season.

Musselman *et al* (1988) examined a number of different ozone exposure statistics, and concluded that no single one was best for all crops. The statistics included:-
- Threshold levels (although the threshold varied with crop)
- Average daily peak concentration
- Concentrations during specific times of the day.

Daily hours above a critical concentration related well to crop yield for alfalfa, tomatoes and dry beans. Average of all daily peak ozone concentrations was the best measure to predict cotton yield.

Lee *et al* (1988) also discussed the problem of different types of measure of ozone exposure e.g. NCLAN 7-h seasonal mean for ambient and experimental exposures. However, peak concentrations have an important effect, and chronic effects may be the result of a cumulative impact of multiple peaks. Acute concentrations and exposure duration are not addressed in long-term averages. Rawlings and Cure (1985) developed the Weibull non-linear function to characterise yield response to ozone pollution. Dassel and Rawlings (1988) determined optimum design point allocation for estimation of relative yield losses: the optimum design depended on parameters used in Weibull equation. Olszyk *et al* (1988) studied the impact of 8 possible scenarios on 20 crops. These included a 1-h oxidant standard of 100 ppb (196 µg m^{-3}), mean concentrations for ozone for 12-h days and other models. Losses were based on a comparison with a background filtered air concentration of 25 or 27 ppb for 12 or 7 h respectively.

Lefohn (1992) gives a clear explanation of the use of the Weibull function. NCLAN studies often showed a non-linear response to which Weibull curves were fitted (although sometimes polynomial regressions were used). The Weibull model relates yield to ozone concentration by:

y=a exp(-(x/b)c)

where a is maximum yield at zero ozone; b is seasonal mean ozone concentration at which a is reduced by 63%, c is a dimensionless shape parameter, and x is the seasonal mean ozone concentration in ppm. It fits most responses observed in NCLAN studies (except those showing greater yield at any ozone treatment above the lowest concentration) and allows direct comparison of ozone effects on yield across all experiments in terms of a proportional yield response when the ‘a’ term is set at 100 for all separate models. Simple linear functions may be used, but do not reflect the curvilinear nature of responses in most crops. He surmised that a combination of Weibull and indices such as SUM06 is the most useful means to predict the impact of ozone pollution on crop yields.

Lefohn and Foley (1992) showed that repeated measures of hourly average ozone concentrations above 100 ppb adversely affect vegetation. Dassel and Rawlings (1988) used Weibull functions with mathematical design strategies to protect against incorrectly guessing Weibull model parameters, when the true values are not known. Lefohn, Laurence and Kohut (1988) compared a number of indices to describe the relationship between exposure to ozone and reduction in agricultural yield. Hourly mean concentration data were used to develop indices of exposure. They looked at the comparative efficacy of;

Cumulative indices... a. No. of occurrences equal to or above a selected level
b. No. of occurrences equal to or above a specific hourly mean concentrations

Means calculated over an experimental period to describe the relationship between exposure to ozone and reductions in yield. None of the exposure indices tested consistently provided a best fit either with Weibull or linear models.

Lefohn and Runeckles (1987) looked at different standards for ozone exposure against dose considerations. Plants are sensitive to different hourly mean ozone distribution patterns, even though seasonal means may be the same. Thus a long-term ozone standard will not protect vegetation from repeated peaks. They make reference to NCLAN data on beans, tobacco, alfalfa and *Pinus strobus*.

Rawlings *et al* (1988) published another critique of statistical approaches, comparing Weibull curves with polynomial and linear regressions etc. Hogsett *et al* (1987) examined exposure indices including:

Mean hourly concentrations, including 7 and 12-h means over the growing season
Cumulative approach
One time event approach
Concentration-weighting approach
Multicomponent indices
Dose approach

Their evaluation suggested that the best indices were cumulative and weighted for the higher concentrations.

14. EFFECTS OF WATER USE AND POLLUTION

14.1 Introduction

Fuel cycles have many different effects on water resources. In Western Europe the most notable effect is freshwater acidification, which has been a major catalyst in negotiations for reduction of emissions of the main trans-boundary air pollutants.

There are also a number of other burdens that affect water resources and associated ecosystems, though these typically act on a more restricted, local, range. They include:
- Emission of polluted water from boiler purge, FGD waste treatment, etc.
- Thermal emissions
- Consumption of water, particularly by power plant cooling systems
- Entrapment of fish and other organisms in cooling water intakes
- Contamination of surface and ground water as a consequence of rain running off and draining through solid waste

The ExternE Project has typically assumed that these more localised impacts, affecting aquatic and riverside ecology and the quality of drinking water, are negligible in the context of analysis of power plants in Western Europe. There are good grounds for assuming that this is often (though almost undoubtedly not always) the case. Firstly, the scale of action is local and hence there is a strong limit on the number of people and ecosystems that are likely to be affected. [Of course, the establishment of a chemical or thermal barrier across a river, although localised, could be expected to have impacts much further up- and down-stream.] Secondly, and most importantly, the current status of environmental law in the European Union with emphasis on Environmental Impact Assessment (EIA), should identify in advance the possibility of environmental harm arising from a project, which can then be averted in the planning process. Clearly this assumes that the EIA process is conducted thoroughly and well.

We note that a more detailed assessment of water-related issues would be beneficial. However, for the present we restrict ourselves to assessment of the impacts related to freshwater acidification (Section 14.2). Section 14.3 summarises the valuation literature on water resources and pollution. So far these data have been applied in a very limited way across the project.

14.2 Quantifying the Impacts of Freshwater Acidification

14.2.1 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 is 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 that 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 14.1. 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.

An additional factor should 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, much of which is non-Scandinavian in origin (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.

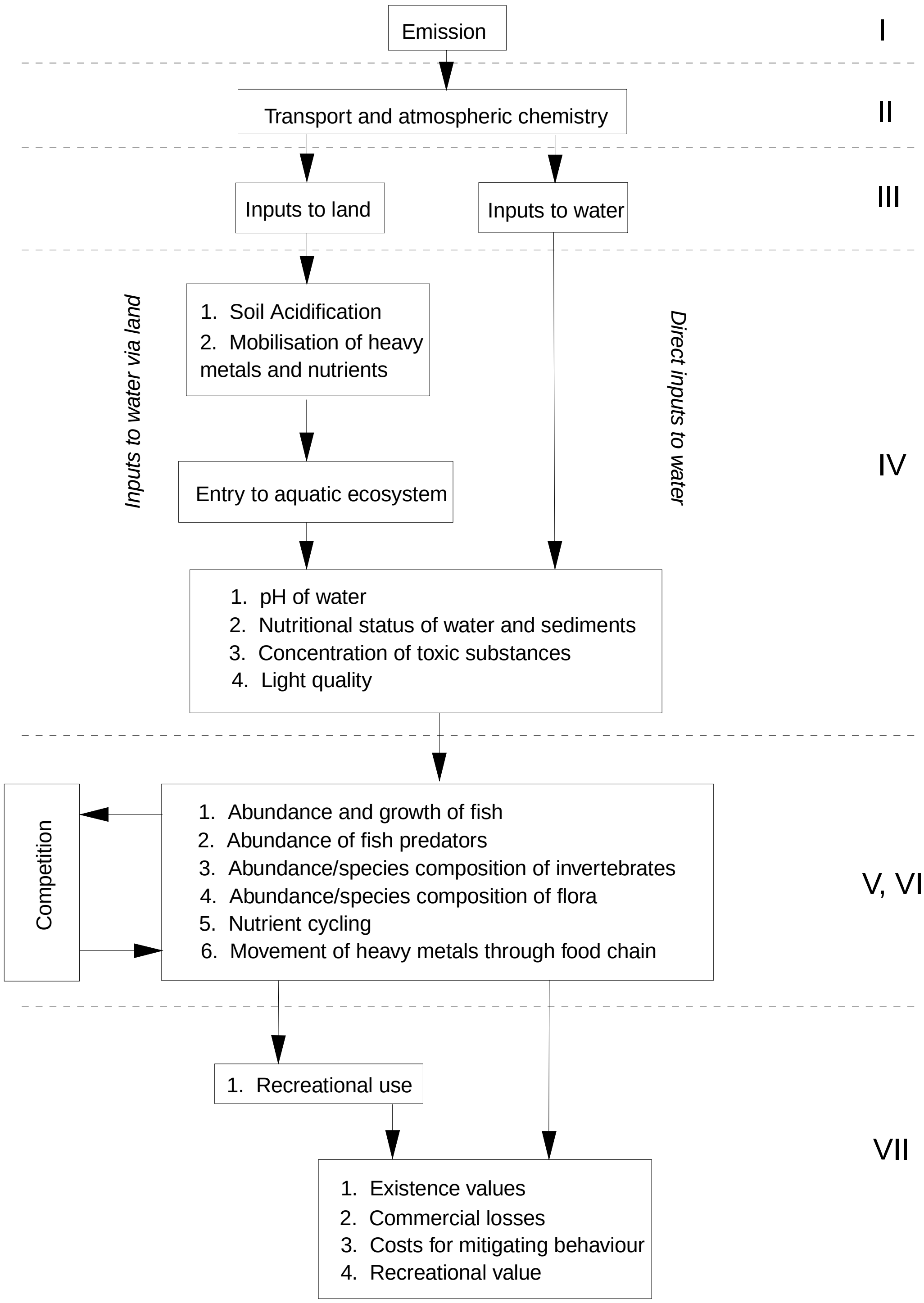

Figure 14.1 Impact pathway for acidic deposition effects on freshwater fisheries.

14.2.2 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.

14.2.2.1 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 that 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, and 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.

14.2.2.2 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 14.2.

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.

14.2.2.3 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:

$$\mathrm{Log}_{10}\, D = -1.24 - 1.08[\mathrm{Log}_{10}\, \mathrm{Al}_{filt}] + 1.33\mathrm{Log}_{10}\, \mathrm{Hardness} - 0.22\mathrm{Log}_{10}\, \mathrm{ADF}$$

where;
D = density as number of fish / 100 m^2,
Al_{filt} = Aluminium in mg l^{-1},
Hardness = mg $CaCO_3$ l^{-1},
ADF = average daily flow in m^3s^{-1}.

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 has been explored in a Norwegian context by Cosby *et al* (1994).

14.2.2.4 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 14.2). 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.

14.2.2.5 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 that 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.

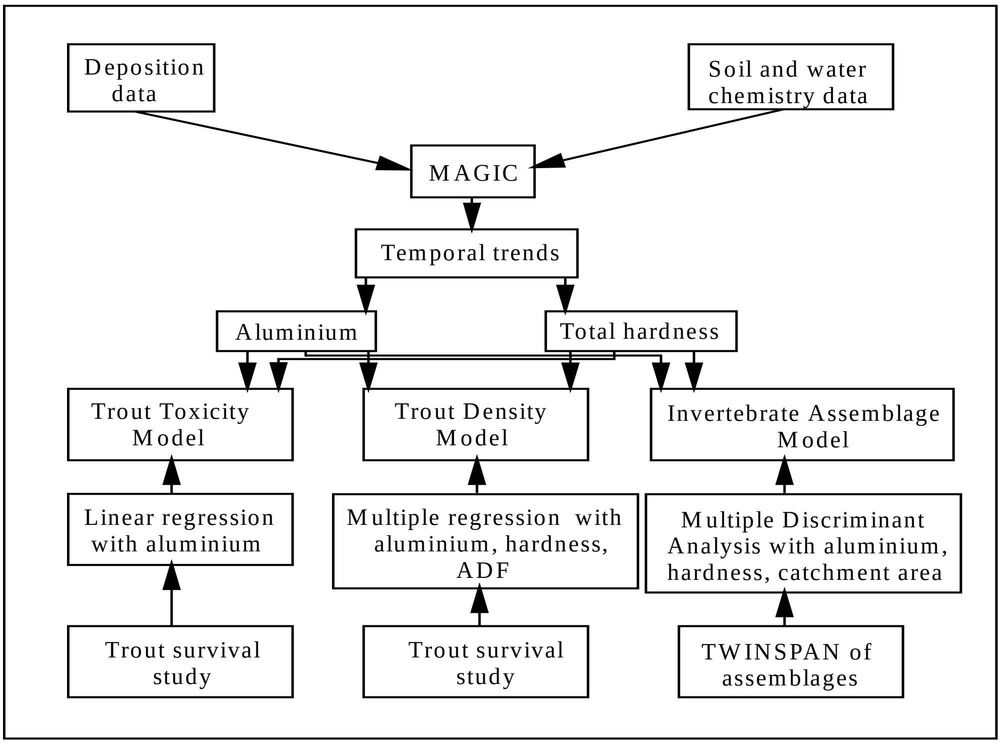

Figure 14.2 A schematic representation of the modelling procedure applied to Welsh catchments (after Ormerod *et al*, 1990).

The fish density model is a statistical model that 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.

14.2.2.6 Demonstration of the methodology

The application of the approach described here was demonstrated in the ExternE coal report (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.

14.2.3 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 can be used to estimate the incremental amount of acidity that would be deposited in Norway and Sweden from a given country's, or power plant's emissions (European Commission, 1995a, b). This is expressed as a fraction of the total deposition in Norway and Sweden (separately). The result is 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 are probably underestimated by this method, as high level emissions travel further than low level emissions.

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.

14.2.4 Summary of Analysis of Freshwater Acidification

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 exceedence 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 load exceedence, 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 an 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 exceedence 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.

14.3 Valuation of Damages to Water Systems

14.3.1 Introduction

This impact category probably has the largest valuation literature but most of it is not directly applicable to the valuation of those damages that might be caused by fuel cycles. Recreational benefit estimation has been studied the most in the context of water. The travel cost method is still the most prevalent method for obtaining recreational benefits although contingent valuation has also been used frequently in conjunction with travel cost.

The external effects of water have been divided into the following sub-categories, each of which is considered below, noting that nearly all the studies have looked at the effects of nutrient emissions (e.g. nitrogen and phosphorus) rather than fuel cycle emissions.

1. Surface Water
    - a) Non-ecological use
    - b) Ecological use (i.e. water-based recreation, except angling)
    - c) Ecological non-use
2. Ground Water
    - a) Non-ecological use
    - b) Ecological non-use
3. Commercial Fisheries
4. Recreational Fisheries
5. Non - Use Fisheries

14.4 Issues

There are many difficulties in using existing water amenity valuation studies to estimate fuel cycle impacts. First and foremost, one has to discount virtually all the studies that only value a scarce resource, e.g. the fishing value of a lake or river. It will very rarely be the case that the activity of a fuel cycle will result in the elimination of the entire amenity. Hence one has to focus on studies that give an estimate of what the marginal value of a resource is. Second, there functions for different pollutants (and range of concentrations), different species, hydrological and climatic conditions. Unfortunately the European valuation studies that do estimate losses in fisheries do not supply the information on the reductions in pollution that they are assuming. In the absence of such information it is not possible to calculate an average loss per ton of pollutant, which would be essential in any fuel cycle study. It may be possible to recover the data by going back to the original authors. If so, such a retrospective exercise may well be worth carrying out.

14.4.1 Results

14.4.1.1 Surface Water: Non Ecological Use

The following studies are relevant:

- Baan (1983)
- SPCA (1991)
- Winje et al (1991)
- Ewers and Schultz (1982)

Baan (1983), studied the social benefit in the Netherlands resulting from higher surface water quality. The SPCA (State Pollution Control Authority) (1991) made a study in the North Sea as to the cost effectiveness of different measures to achieve reductions of pollutants. This involved a 50% reduction in national emissions of Nitrogen and Phosphorus to the North Sea. The discounted costs (discount rate = 7%) of different measures for reducing emissions of Nitrogen and Phosphorus by 1 tonne were NOK 0.02, 0.15, 6.95 and 30.90 (1990) respectively (1 NOK = 0.125 ECU). The basic data on the relationship between costs and reductions of pollutants could be of relevance in deriving a response to any estimated damages from the pollutants, but cannot in general be a substitute for estimating those damages in the first place. Hence this study is not of direct relevance to the ExternE Project as a whole.

In a similar vein, Winje et al (1991) in West Germany looked at the costs of treating different pollutants in 3 sectors including public drinking water supply, private and industrial water supply. Pesticide residuals (PPP), chlorinated Hydrocarbons (CHC) and nitrate concentrations were investigated. Treatment costs for industry = DM 120 million per annum (1983) and public supply = DM 780 million per annum (1983). 60% of these additional future costs are due to nitrate concentrations in ground water.

Ewers and Schultz (1982) similarly value the benefits of improved water quality in terms of the reduced cost of treating it. They look at the value of improved water quality (reduced eutrophication due to reduced emissions of phosphorous) in Lake Tegeler, Germany. For drinking water quality improvements (see later for the same study valuing commercial fisheries) from the present level of quality (level 4) up to the highest level (level 1) were valued in the range DM 4.7-6.9 million annually. For catering firms the corresponding estimate was DM 0.4-4 million. The drinking water benefits were valued by the associated cost reductions of improved water quality.

Although such a procedure is not generally valid as an estimate of the damages from a polluting source, it *can* be a measure of the benefits if it can be established that the water treatment measures *would have to be taken on grounds of public health, or if the damages would be so high that the treatment costs would be much less than any damages*. In general this is not known, and it is one of the aims of this study to compare the costs and benefits of pollution control. In the exceptional cases, however, where it is valid to do so, the estimate of the costs from the three studies quoted above would have to be converted into a cost per unit of pollution. This will require data on the pollution levels in quantitative terms, which are not available from the published data.

14.4.1.2 Surface Water: Ecological Use

The following studies are relevant:

- Hjalte *et al* (1982)
- Kanerva & Matikainen (1972)
- Green and Tunstall (1990a/b); WRC/FHRC (1989)
- Magnussen & Navrud (1991)/Magnussen (1991)
- Hervik, Risnes & Strand (1987); Strand & Wenstop (1991).
- Mäntymaa (1991)
- Aarskog (1988)
- Heiberg & Hem (1988)
- Dalgard (1989)

Hjalte *et al* (1982) in Lake Vombsjon, Southern Sweden (drinking water source of the town called Malmo) estimated the recreational value per visitor per year at SEK 4 (1982), on average for all recreational activities. WTP = SEK 4.50 for angling alone. (1 SEK = ECU 0.13). Variables measuring water quality were in terms of angling, bathing, boating, bird watching, walking and lakeside view. This study used a theoretically acceptable travel cost model, e.g. taking account of substitute sites, varying on-site costs and not over stating time costs (i.e. use 15% of the average industrial working salary). Three different water quality levels were examined in qualitative terms. In these circumstances the damage function would have to be discontinuous at the three points indicated by the three levels of the quality. If an energy source were to cause the water to fall from one quality level to another, the estimated damages would be measured in terms of the loss of that specific function. There is no reason why data of this type should not be used in fuel cycle studies, but it will require very site specific estimates of the impact of pollutants from the plant on the local water quality.

Magnussen & Navrud (1991) / Magnussen (1991) is probably the most comprehensive Norwegian study on user and non-user benefits from water quality changes because it considers both the linkages to physical damage functions and the constructed CVM models. Water quality was found to be mainly affected by Nitrogen and Phosphorus emissions and the study looked at the benefits of a 50% reduction in emissions. The average WTP per household for the improvement was estimated at NOK 600 - 5000 1991 (72-600 ECU, 1990). Again, this study only measures large changes in water quality so that the valuation procedure in which it is used would have to be adapted accordingly.

Another study, which could be relevant to the fuel cycles project in that it considers the effect of diminished water quality, is Mäntymaa (1991). Again, it operates in terms of 5 quality levels. The task of linking specific pollutants to these quality levels remains, however, incomplete, or at least not available from the published version.

The unit values estimated in Green and Tunstall (1990a/b) / WRC/FHRC (1989) can only act as a rough indicator of the valuations in the UK that people place on water quality improvements. Problems in using these studies stem from the fact that people do not perceive the difference between the present water quality levels and those water quality improvements that are being valued. It is thus very difficult to place much reliance on the results and to transfer these results to the fuel cycle external cost study.

Table 14.1 shows the findings of three main Norwegian studies. The physical changes measured in these studies (i.e. oxygen concentrations, eutrophication, heavy metal and chlorinated hydrocarbon concentrations and surface pollution of oil and litter) were 'translated' into perceivable measures of water quality but the water quality improvements were rather large, from the prevailing state to a 'nearly unpolluted' state. Thus, although the improvement is very large, both steps required to relate the results to a fuel cycle study are available - i.e. the link between the pollutants and the measures of water quality, and the relationship between water quality and WTP.

Table 14.1 Mean annual willingness to pay per local household for improved water quality in three Norwegian fjords.

Fjord	Author[1]	Mean WTP per household per year		
		Users	Non-Users	Weighted average of Users and Non Users
Kristiansand	Heiberg & (1987)			430 (62.5)
Inner Oslo	Aarskog (1988) / Heiberg & Hem (1988)	906 (123.4)	598 (81.4)	837 (114.0)
Drammen	Dalgard (1989)	849 (110.5)	416 (54.1)	563 73.3)

1. All studies have used contingent valuation mail surveys using colour coded maps and detailed verbal descriptions of different pollutants before and after the improvement. All figures are in NOK; except the WTP figures in brackets which are in ECU, 1990.

For recreational use values, results are often reported in terms of the value of a recreational day. The above studies do not, unfortunately, permit an easy calculation of such a value, but one has been provided by Navrud (1994). He estimates, from a range of Scandinavian and US studies the following values for different recreational days: Atlantic salmon/sea trout fishing (ECU 16.3); brown trout fishing (ECU 7.6); big game hunting (ECU 20.6); small game hunting (ECU 16.4); hiking and cross country skiing (ECU 5.4); picking mushrooms (ECU 8.1); and swimming (ECU 4.3). To test the transferability of these estimates he examined an EIA for a proposed hydropower site in Sauda Fjord in SouthWest Norway. Losses of fish and wildlife stocks had been estimated in the EIA, and 'expert opinion" was used to convert these into lost recreational days. Taking the above values for the different recreational days he obtained an estimate of total loss due to the hydropower development of ECU 142,000/year. He then conducted a direct CVM of the potential users, focusing on the loss of recreational value in the area. By this means he obtained an estimate six times larger (ECU 809,000/year).

Navrud's analysis is interesting but not conclusive. It strongly suggests that marginal valuations made up of benefit transfer and expert assessments are misleading. However, it does not tell how transferable a 'proper' marginal valuation study would be. For example, would the results of the specific study be convertible into value per 1000 loss of fish population and then be transferred to a new situation? Certainly such an issue is worth exploring.

14.4.1.3 Surface water: Ecological non-use

The following studies are relevant:

- Kyber (1981)
- Magnussen & Navrud (1991)/ Magnussen (1991)
- Hervik, Risnes & Strand (1987); Strand and Wenstop (1991).
- Mäntymaa (1991)
- Green and Tunstall 1990a/b; WRC/FHRC (1989)
- Aarskog (1988)
- Heiberg & Hem (1988)
- Dalgard (1989)

All the studies presented in Table 14.1 had a low response rate (40%-50%) and the number of respondents varied between 300 and 400. However, tests of the samples' representativeness with respect to age, sex and income indicated that the samples were indeed quite representative. In the Kristiansand fjord study a national CVM survey yielded a national (single payment only) WTP of NOK 1515 million (1990). Unfortunately, nothing was done to counter any possible part-whole bias or focusing effect that would overestimate the valuation.

Kyber (1981) made a hedonic study of the water quality in Valkaekosia, Finland affecting average value per square metre of shore area with summer cottages and permanent dwellings. He used the National Board of Waters' quality levels where quality level II indicates 'good' quality; level III (satisfactory); IV (passing) and level V (poor). Valuations were found but no details were given regarding the levels of physical pollutants and the equivalent measures used by the national water board. Since water quality in this sense is only weakly related to pollution levels, a link between the two might be very difficult to establish. A similar comment applies to the study by Mäntymaa (1991) of Lake Oulujarvi, Finland.

Green and Tunstall 0 TD valuah(i)-2.7(nl7a0 TDT972.7valuah(Tc0.011 2.7u)-4(e)11174.9(sl90 TDT978(eu9p0

The mean WTP per person per annum was; ECU 814 for class 1, ECU 786 for class 2 and ECU 737 for class 3. The small differences in values are remarkable, even for non-users one would expect bigger values and for users, US studies suggest much bigger differences. The finding was, however, supported by a survey of all user benefits where the recreational values per visit were ECU 0.17, ECU 0.84 and ECU 0.73 for water quality levels, 3, 2 and 1 respectively. The typical present value of the benefits of improving the water quality level from one class to the next is as follows:

Class 3 to Class 2 = £450,000 (for a country park)
Class 2 to Class 1 = £90,000 (for a local park site)

These results are too general to be of direct use in a fuel cycle study. If one could establish that the operation of a particular fuel cycle resulted in a change in quality as identified above, however, it might be possible to use this and similar studies to value the impact.

Hervik, Risnes and Strand (1987); Strand and Wenstop (1991) established a WTP of NOK 550 - 1490 (1990) through a CVM survey for the preservation of the River Rauma-Ulvaa, West Norway. The study was based on the most cost-effective way of ranking rivers for hydro-electric development. This may be relevant to the external costs of hydro power in valuing the non-use value of the site prior to development.

Methodologically, it is difficult to separate the use and non-use values as clearly as one might like. Where genuine non-use values are involved, there will always be a problem of relating such valuation to one or more stages of the fuel cycle. By their very nature non-use values are not closely or quantitatively linked with pollution levels. If preservation is threatened, the relevant loss of value is that associated with the affected habitat or environment and some of the preservation valuations that have been carried out may be of relevance, although it will be important to ensure that the results are broadly transferable. In other cases, such as those where individuals enjoy the visual impacts of a clean lake or object to the idea that a water body is being polluted even when they will make no use of it, valuations *may* involve a relationship between the level of pollution and the WTP. However, this is very difficult to measure and the best that can be hoped for is a specific CVM study when the main source of the pollution damage is a specific power plant. Naturally such a study would be specific to that plant, although it may be possible to transfer some of its results to other plants in similar circumstances.

14.4.1.4 Ground Water

The following studies are relevant:

- Silvander (1991)
- Hervik and Strand (1991)
- Hanley (1989)
- Winje et al (1991)- see surface water ecological non-use
- Hübler et al (1991)

Most of the studies in this area relate to nitrate leaching and its impact on ground water. Silvander (1991) estimates the social benefit of reducing nitrate concentrations in groundwater to be approximately SEK 5 - 51 (ECU 0.65 - 6.6) per Kg of nitrogen leached. This includes use and non-use values but the separation of the two is not possible from the published version.

Strand and Hervik (1991) examined user interests, as affected by energy generation per unit, to maintain a river in Norway. The main results are shown in the annex. The WTP of different user interests is derived from implicit decisions of policy makers, so that a decision to undertake development A rather than B reveals certain priorities about the relative environmental costs of A versus B, given that the non-environmental costs are known. The method of this study is unorthodox but interesting. A basic assumption for such implicit valuations to be valid is that policy makers are well aware and informed of the environment/economy trade-offs involved. Whilst this assumption seems to be fulfilled in this case, it may not often be the case in other similar studies. This study is relevant to hydroelectric power and may be of great importance in valuing the hydro cycle. Results are likely to be country specific, since polices towards hydro development vary between countries.

Winje et al (1991) have given their results as a total cost (estimating avoidance and transaction costs of treating pesticide residuals, chlorinated hydrocarbons and nitrates) for both surface and ground water. The treatment costs cannot be calculated for ground water separately. The earlier comments about the use of avoidance costs apply here as well.

The final report for the Hübler et al (1991) study is not yet published and the national estimates are based on available monetary estimates of a few polluted areas. The aggregation is based on non-monetary information and expert judgements. However, data on the polluted areas could be used to provide unit values if they are detailed enough to give values for different pollutants.

Hanley (1989) conducted a CVM survey in the East Anglia region of the UK in an attempt to establish a WTP for a water quality below the WHO threshold for nitrate concentrations (i.e. less than 50ml/l). He obtained an estimate of £13 (ECU 20.0) per person per year using a payment method of increased water rates. The author noted that public knowledge regarding the general problem of nitrates was acceptable although it may be difficult for respondents to perceive marginal changes in nitrate concentrations of ground water.

A very similar study was conducted by Silvander (1991) in Sweden. He looked at the surveyed WTP to decrease risk of methemoglobinemia in infants. This question yielded an annual WTP of SEK 244 (ECU 31.7) per person. Reducing the risk of cancer in adults and infants gave an annual WTP of SEK 347 (45 ECU, 1990) per person. Total WTP for both health effects per person is SEK 591 (ECU 76.8) and SEK 522 (ECU 67.9) for those that did not know about the nitrate levels and those that did, respectively. The corresponding average WTP per person per year is SEK 332 (ECU 43.1) and SEK 190 (ECU 24.7) respectively. 84% of respondents did not know if they were subject to nitrate concentrations above the threshold. The payment vehicle is increased income tax. A serious problem with this study is that reduced risk for the two diseases is not described in detail. If this could be provided, however, these results could be of potential use in a fuel cycle study where it could be established that nitrate concentrations were affected relative to threshold levels by the activities of a power plant.

14.4.1.5 Commercial Fisheries

The following studies here have a limited use for estimating marginal damages:

- Ewers and Schulz (1982)
- Baan (1983) (see earlier section on Water - non-ecological use of surface water).
- Rasmussen *et al* (1991)

Baan (1983) used unadjusted market prices and unreliable dose-response functions to estimate the value of the loss to commercial fisheries and hence his results cannot be relied on.

Ewers and Schulz (1982) in a study on the Lake Tegeler, West Germany (see also non-ecological use of surface water) measured the level of commercial catch as affected by reduced phosphorus emissions. The pollution levels used in the study made no reference to actual physical measures of phosphorus. The authors only mentioned an annual producer surplus increase of DM 12,400-DM 44,000, which they said was gained if pollution improved from level 4 up to level 1 (cleanest). Thus, the link between the phosphorous levels and the water quality indicators needs to be established.

The study by Rasmussen *et al* (1991) includes the effects on deep sea fishing; coastal fishing in the North and Baltic Seas; river fishing; lake fishing and aquaculture (ponds) in West Germany over the period 1950 - 1982. He estimated an annual loss of DFL 130 million (ECU 56 million, 1987). Total social costs were estimated. This involved taking into account the fact that

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Table 14.2 Review of European studies on the recreational value of freshwater and saltwater angling.

River	Species	Author	Method	Recreation value per fishing day (1990 NOK)
R.Gaula	salmon/trout	Strand (1981)	TCM	ca.325
		Rolfsen	TCM	424 - 584
		(1990)	CVM	309
		Singsås (1991)	TCM	209 - 326
R.Vikedal-selv	salmon/trout (acidified)	Navrud (1988)	TCM	134 - 183
			CVM	126 - 180
R.Audna	salmon/trout (acidified)	Navrud (1990)	TCM	206 - 234
			CVM	90 - 263
R.Stor-dalselv	salmon/ sea trout	Ulleberg (1988)	TCM	226 - 299
R.Halling-dalselv	Brown trout	Navrud (1984)	TCM	ca.165
R.Tinnelv		Scancke (1984)	TCM	ca.165
Lake Lauvann		Navrud (1991a)	TCM	114 - 145
			CVM	73 - 99
Gjerstad-skog			TCM	82 - 91
Lakes			CVM	42 - 63
Sea area near R. Audna	Salmon/ sea trout	Navrud (1991b)	TCM	ca. 55
			CVM	40 - 65
All Swedish salt and brackish water	Most salt-water species	Silvander (1991)	CVM	ca. 25

NOTE
TCM = Travel Cost Method — 1 NOK = 0.125 ECU, 1990.
CVM = Contingent Valuation Method — N.B Both the TC and CVM methods yield similar results.

From the results of the studies in Table 14.2, it should be possible to construct a *willingness to pay function* of how recreational value per angling day varies with the characteristics/quality of the water body (e.g. salt or fresh water, lake or river and the particular fish species present), availability of substitute sites and demographic/socio-economic variables of the regional population and anglers. All these variables may be different for each European country yet if those differences can be identified and measured it should be possible to transfer the willingness to pay function to other countries. This would constitute a meta analysis of the kind described in Section A of this Report. Including in the function variables such as water quality would then permit the valuation of recreational losses in a specific fuel cycle study where pollutants could be linked to water quality indicators. However, such a meta analysis remains to be done.

Another study that could be of direct relevance is that of recreational fishing demands and water quality, carried out by Navrud (1988) in Norway. He examined the water quality affecting fish (Brown Trout stock) populations from 1980. Respondents were asked to reveal their WTP for various intensities of lime application to achieve 30%, 50% and 70% reductions of sulphur effects at all sites. Respondents were also questioned regarding quality of all public goods affected by acid rain. This was used to control against 'constant budget bias', but the results are themselves useful in indicating the WTP for an improvement in water quality that has been damaged by acid rain. The results are summarised in Tables 14.3 and 14.4.

Table 14.3 Maximum annual WTP of Norwegian households for increased freshwater fish populations [WTP-Fish] from reduced European sulphur emissions (1986 NOK)

Sub-sample Number (emission reductions)	No. of observations	Response rate Fish/Public goods	First Bid	WTP - Fish/HH/year		
				Mean	Med	Std Dev
1	288	98/97	200	278	200	338
2	270	97/95	500	455	200	826
3	238	99/96	1000	603	300	772
4	239	97/99	0 - 10000 (pc)	335	100	514
5	206	99/96	200	366	200	660
6	206	98/98	500	578	200	1036
7	204	97/97	1000	597	200	796
8	192	98/98	0 - 10000 (pc)	291	100	492
9	189	98/97	0 - 10000 (pc)	387	200	648

Sub-samples 1-4 were asked for a WTP to reduce sulphur by 30%
Sub-samples 5-8 were asked for a WTP to reduce sulphur by 50%
Sub-sample no. 9 was asked for a WTP to reduce sulphur by 70%

(pc) = payment card, where 3 of the sub-samples were shown amounts ranging from 0 to NOK 10,100 and asked to pick the value that reflected their WTP.

Table 14.4 Maximum annual WTP of Norwegian households for all public goods [WTP-Public Goods] from reduced European sulphur emissions (1986-NOK)

Sub-sample Number (emission reductions)	WTP-Public goods/HH/year			Mean WTP-Fish Mean WTP-Public Goods	WTP-Fish: zero bids
	Mean	Med	Std Dev		
1	375	200	652	74	23
2	537	200	1003	85	18
3	700	250	1285	86	23
4	478	100	1130	70	29
5	617	200	1324	59	23
6	793	300	1883	73	17
7	656	200	962	91	23
8	444	150	998	66	28
9	387	200	648	64	27

Average WTP for all of Norway = 300 NOK (37.5 ECU, 1990) per annum [1 NOK = 0.125 ECU, 1990). Payment cards and bidding games used to avoid instrumental bias. Significant variables included the method of payment, income level, age, use of facility, sex and location.

This study could be of potential value in the ExternE study as it contains marginal valuations of sulphur emissions (Table 14.3), both in terms of NOK/fish/year as well as NOK/HH/year. It also provided careful estimates of changes in fish stocks, based on dose-response functions, something which other studies do not always do. However, it is worrying to note, in spite of this careful estimation of impacts he finds, as with the Travers Morgan study, that there is very little increase in WTP for increased sulphur reductions. Given the large expected physical impacts of such reductions, it suggests that there is still a problem in eliciting marginal valuations.

Finally there is a study of the value of extending the fishing season and the length of the river with fishing rights. Bonnieux, Desaigues and Vermersch (1990) used the CVM method and found a mean WTP of ECU 14.9 to increase the duration of the fishing season by 25%. This monetary amount is equivalent to a 25% increase in the present fee charged for the use of the salmon lakes.

The authors found, however, that knowledge regarding fish stocks was inconsistent and some anglers thought stocks were severely depleted despite information given in the survey. This indicates that anglers are WTP 25% more if they can fish 25% longer. A mean WTP of ECU 83.8 was also established to buy a further length of river corridor of 5 km. Only 40 anglers were WTP an average of ECU 82.2 for yet another 5 km. Thus the first 5 km is valued at ECU 3803.8 per km per year. The second 5 km is valued at ECU 2921.5. Other important factors influencing the results included income level, and those living far will have a high demand for extra bank

14.4.1.7 Fisheries (non-use)

The following studies are of potential relevance:

- Navrud (1989)
- Navrud (1990a/b)
- Amundsen (1987)
- Carlsen (1985)
- Strand (1981)
- Silvander (1991)

The results are summarised in Table 14.5. All the studies employ the CVM to elicit WTP. With the exception of Silvander (1991) which is Swedish they are all Norwegian. The Silvander study is also the only one to use mail surveys whereas the others conduct in-person interviews (which should give better results). Four of the studies give the total WTP (use and non-use); Navrud (1989), Amundsen (1998), Carlsen (1985), Strand (1981). This is important to avoid double counting. Strand (1981) and Navrud (1989) have divided the use and non-use values, however, the method for doing this is rather *ad hoc*. Hence, one cannot place much reliance on the separate numbers.

The results in Table 14.5 are of direct relevance to the estimation of the costs of acid depositions (see also Chapter 13 also). It is from detailed studies such as those carried out by Navrud that valuations of acid depositions will be obtained and fed into the proper valuation of acid rain damages. Some of the other studies, e.g. Silvander, which looks at eutrophication damages, may not be relevant to fuel cycles. That will depend on whether the fuel cycle's effects on fish are perceived to be the same or different from the effects of eutrophication. In other cases, the estimated damages are related to extinction. They are relevant only when the impact of the fuel cycle can categorically be associated with extinction. That will rarely be the case, but it is not impossible.

Where the estimates have been done on a per household and per person basis, comparisons can be difficult. However, WTP amounts per person can often be rather close to those submitted per household because individuals can often think in terms of a total household budget and thus act as representatives of the entire household.

14.4.2 Recommendations

This Chapter has reviewed the literature on several categories of water damages. A distinction was made between surface water, groundwater and fisheries; and between use and non-use values. Within use values, separate valuations exist *in some case* for ‘ecological’ uses, such as recreation, and for non-ecological uses such as drinking water.

For surface water, valuations exist for non-ecological uses based on the costs of treatment to reduce the phosphorous and nitrogen levels. It has been pointed out that such costs of treatment are not a substitute for actual damage estimates, unless it is obviously the case that treatment is necessary to a given level. In the fuel cycle study it may be necessary to use these values where a power plant is adding these chemicals to the surface water and treatment is deemed necessary. In such cases water purification costs may be used.

Table 14.5 Non-use values and fisheries

The change in fish stocks valued	Cause of change	Author	WTP per year (1991 NOK)
Detailed description of the increased number of trout lakes and salmon rivers with restored stocks on southern Norway (national)	Reduced acidification due to 30-70% reduction in European SO_2 emissions	Navrud (1989)	390#(46.8 ECU) (246-343 [29.5-41.1 ECU] is non-use value; the rest is use value).
Avoiding the extinction of the current salmon and sea trout stocks in River Audna (local)	Stop liming, i.e. the neutralisation of acid depositions	Navrud (1991)	115* (13.8 ECU)
Avoiding the extinction of current trout stocks in the Gjerstadskog lakes (local)	Stop liming, i.e. the neutralisation of acid depositions	Navrud (1991)	46* (5.5 ECU)
Avoiding an unspecified "reduction" of the current trout stocks in the Oslomarka lakes (local)	Not start liming to neutralise acid depositions	Amundsen (1987)	360# (43.2 ECU) (both use and non-use values.
Avoiding "some" and "considerable" reductions in the salmon stock in river Numedalslågen (local)	Different operation schemes of the hydro-power dams	Carlsen (1985)	41-85# (4.9-10.2 ECU) (only 24-25% of the households were WTP, 165-340 [19.8-40.8] per household) (both use and non-use values)
Avoiding extinction of all freshwater fish stocks in Norway. (National)	Acidification	Strand (1981)	1650-2650* [197.8-317.7 ECU] (1000-1600 [120-191.8 ECU] is non-use value; the rest is use value)
[1]Avoiding extinction of the species most popular for consumption, in all Swedish saltwater areas. (National)	Eutrophication due to leaching of nitrogen	Silvander (1991)	310* (37.2 ECU)

Notes: [1]:Saltwater study; #: WTP per household *: WTP per individual. All ECU figures are 1990 where 1 NOK = 0.125 ECU.

Ecological uses of surface water relate to broad categories of water quality and WTP measures are available and, with some care, may be transferable. However, the study by Navrud (1994) casts doubt on even that transferability. If transferability is to be achieved, a function relating pollutant loadings to these broad water quality measures is required. This will have to be done on a plant by plant basis.

For non-use values of surface water, valuations are difficult to obtain. First, it is not always possible to separate use and non-use values and there is a danger of double counting if both are included. Second, non-use values are often for the general preservation of a vaguely defined water body in a certain state. If preservation is clearly threatened, as for example may happen in developing a hydro project, then specific local valuations will have to be made. In other cases, however, marginal impacts from power plants are likely to be immeasurable.

Similar conclusions hold for ground water. For use values, WTP measures are available, as are links between the relevant quality measure and the pollutant discharges. Hence, it is a case of identifying the amounts discharged in the case of the plants being valued.

For commercial fisheries, dose-response functions are poor and, apart from some lake fish impacts the valuation is not reliable. Hence it is recommended that this category of impacts be excluded, unless it is clear that they of great importance, in which case a special effort will be needed.

For recreational fisheries, the work of Navrud is of particular value in estimating the damages caused by acidic deposition. As in the case of forests, one could develop an aggregate measure of damages caused by acid deposition to lakes in each country, and then relate them to emissions from each country. This will require, however, data on the WTP in each country, and so far the bulk of the information is for Scandinavia. The transferability of these estimates must be questionable. It would be worth developing this work, but at the present time, estimates of water damage from acidic deposition will have to remain unvalued.

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15. REFERENCES IMPACTS OF AIR POLLUTION ON BUILDING MATERIALS

15.1 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 discoloration, 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 (see Rabl, 1998).

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 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.

15.2 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 15.1 to 15.4. 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.

15.2.1 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 SO_2 and NO_x emissions. In addition, for some fuel cycles, a simple method has been used to quantify the deposition of particulates on building soiling.

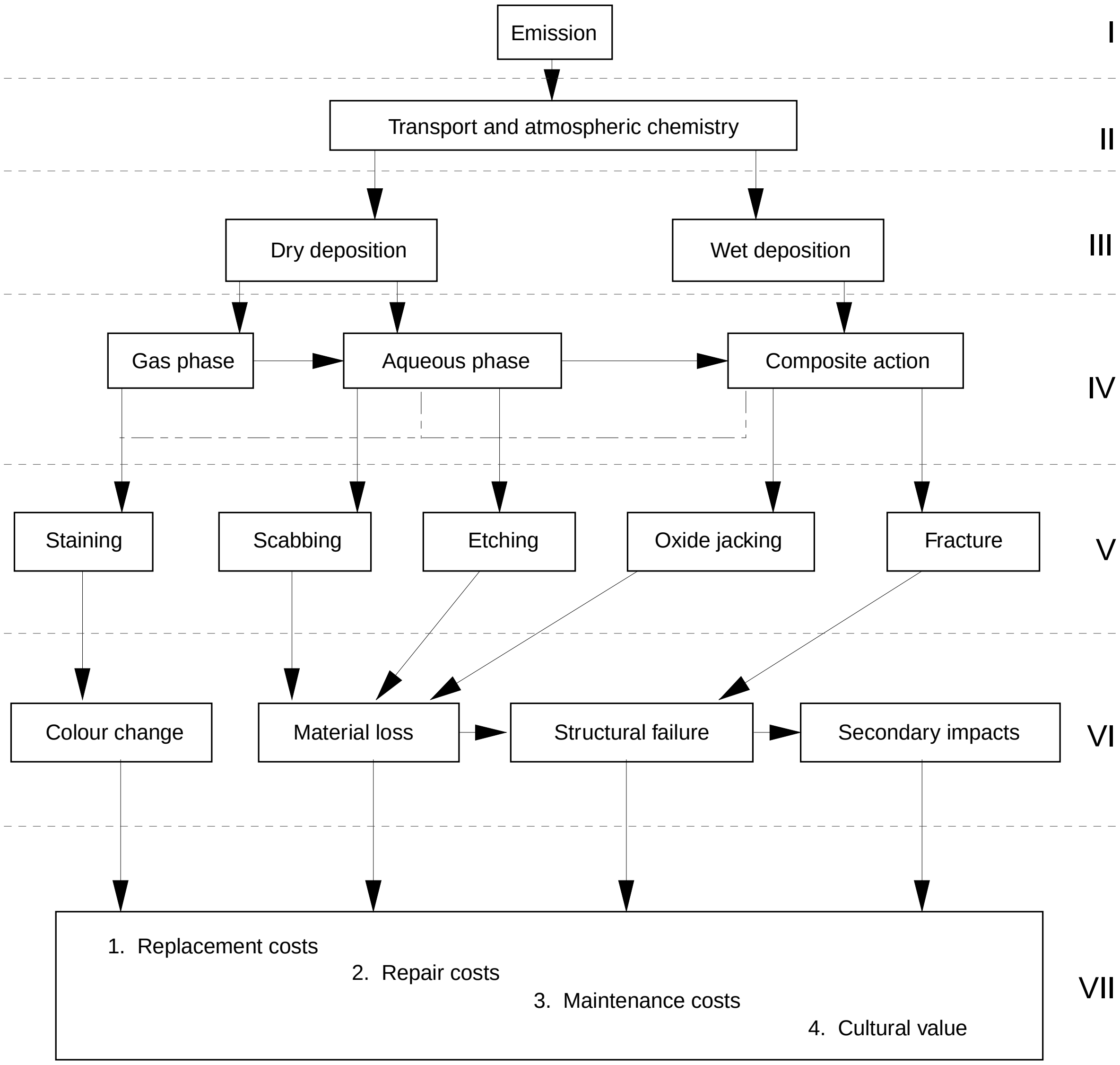

Figure 15.1 Impact Pathway for the Effects of Acidic Deposition on Stone.

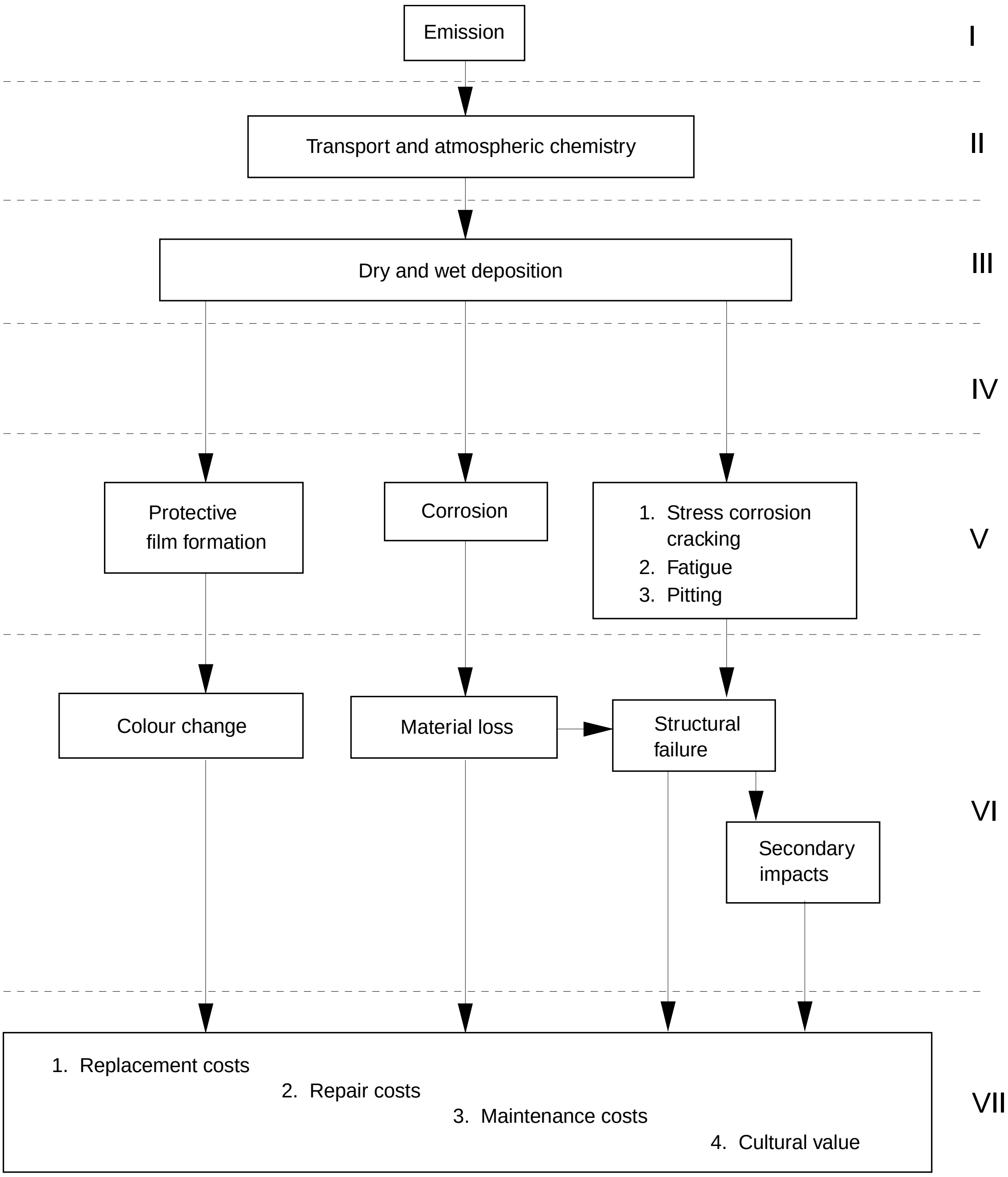

Figure 15.2 Impact Pathway for the Effects of Acidic Deposition on Metals.

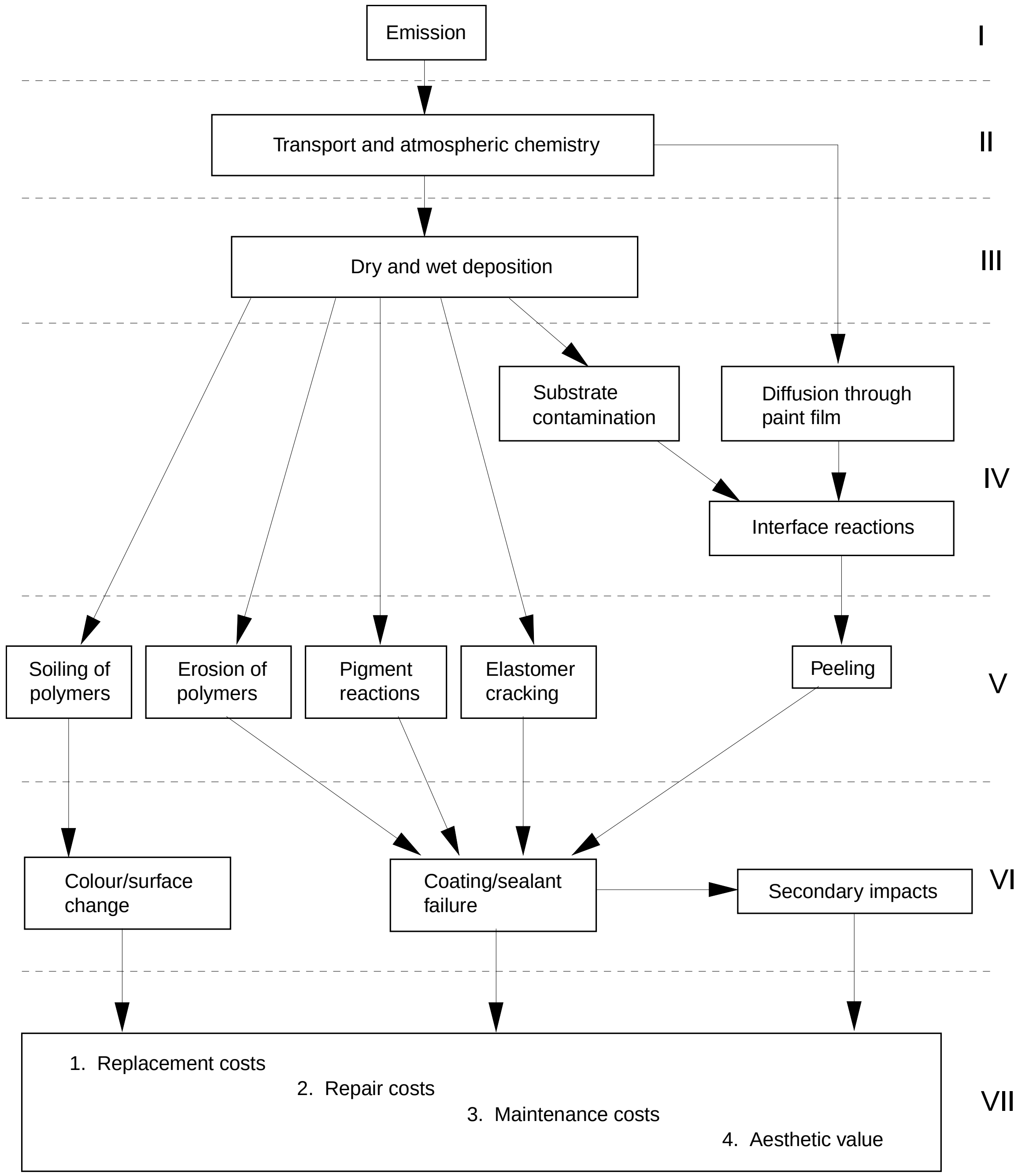

Figure 15.3 Impact Pathway for the Effects of Acidic Deposition on Polymeric Materials (Paints, Plastics and Rubbers).

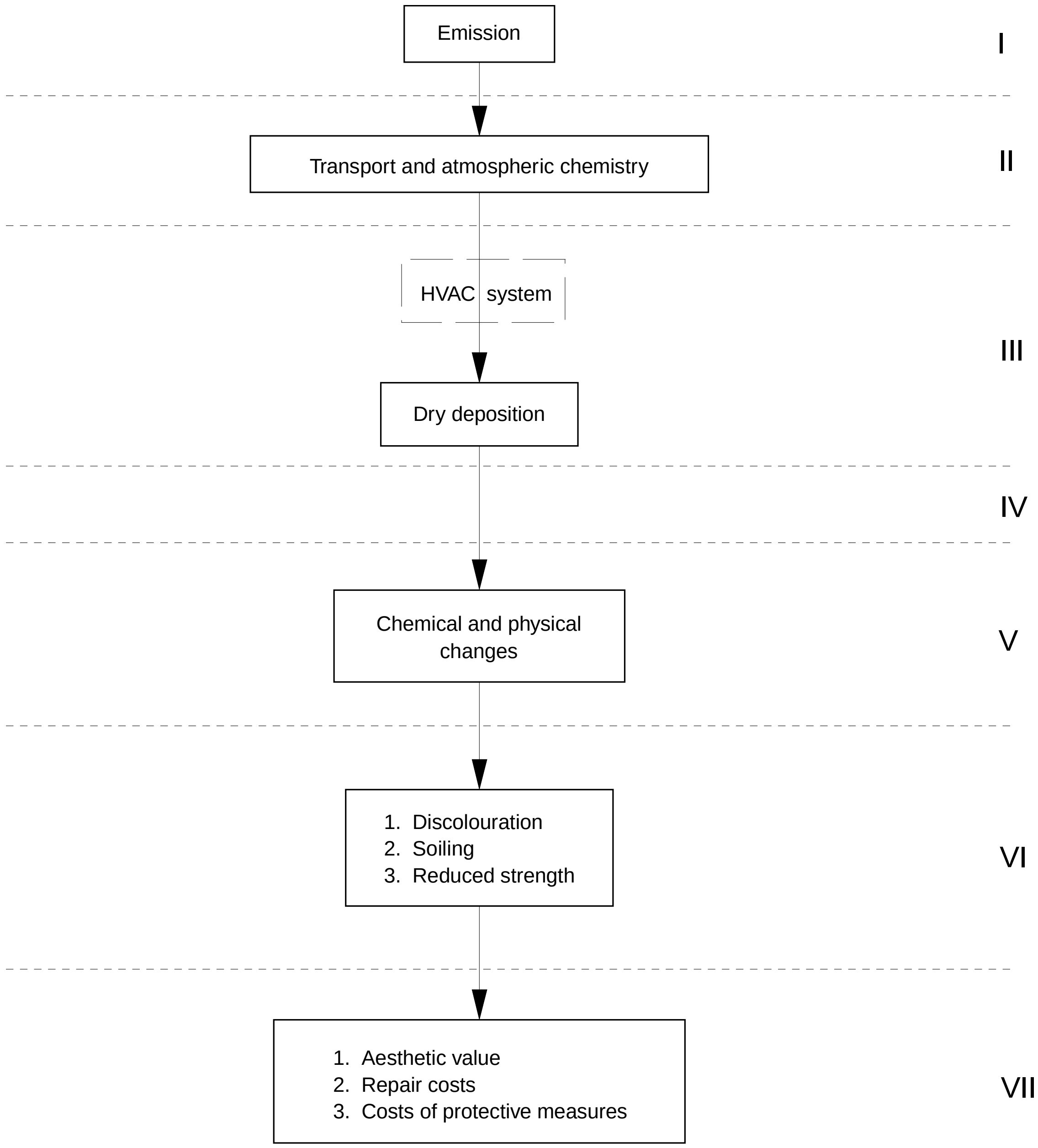

Figure 15.4 Impact Pathway for the Effects of Acidic Deposition and Photo-Oxidants on Fine Art Materials.

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 SO_2 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 NO_2 has not yet been clarified. Although a strong synergistic effect with SO_2 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*, 1996). It has also been observed to act synergistically with SO_2 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.

The materials for which damage has 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 15.1.

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.

Finally, there are impacts from particulate emissions on buildings. The most obvious of these is the discoloration 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. Further possible effects through interaction with other air pollutants are not currently quantifiable.

Table 15.1 Sensitivity of Materials to Air Pollution and the Stock-at-Risk in Europe.

Material	Sensitivity to air pollution	Stock-at-risk in Europe
Brickwork	uncertain	very large
Concrete	low	very large
Natural stone (sandstone, limestone, marble)	high (severely affected by SO_2)	large (especially culturally valuable objects)
Unalloyed steel	high (severely affected by SO_2)	very small
Stainless steel	very low (excellent resistance to pollutant attack)	medium
Nickel and nickel-plated steel	high (especially in SO_2-polluted environment)	very low
Zinc and galvanised steel	high (especially in SO_2-polluted environment)	medium
Aluminium	very low	medium
Copper	low (formation of patina layer in polluted environments)	low
Lead	very low (one of the most resistant materials)	low
Organic Coatings	uncertain	very large

15.3 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.

15.3.1 Regional Databases

15.3.1.1 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. The EcoSense model contains data from a number of such surveys that have been conducted around Europe. Where possible country-specific data have been use. For the most part it is assumed that the distribution of building materials follows the distribution of population. Sources of data are as follows;

Eastern Europe:
Kucera *et al* (1993b), Tolstoy *et al* (1990) - data for Prague
Scandinavia:
Kucera *et al*, 1993b; Tolstoy *et al*, 1990 - data for Stockholm and Sarpsborg
UK, Ireland:
Ecotec (1986), except galvanised steel data, taken from European Commission (1995a); data for UK extrapolated to Ireland
Greece:
NTUA (1997)
Germany, other Western Europe:
Hoos *et al* (1987) - data for Dortmund and Köln

warrant future investigation.

Meteorological, atmospheric and pollution data

The data required to complete the description of the reference environment are the meteorological conditions that 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 $\mu g/m^3$ (Kucera, 1994);

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 O_3 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 SO_2), 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.

Data for other countries were taken from Kucera (1994).

15.4 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 damages 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 types 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, b), and the ICP UN ECE Programme (Kucera, 1994).

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 15.4.

Table 15.4 Comparison of the Dose-Response Functions for Material Damage Assessment.

	Kucera	**Butlin**	**Lipfert**
Exposure time	4 years	2 years	-
Experimental technique	Uniform	Uniform	Meta analysis
Region of measurement	Europe	UK	-
Derivation of relationships	Stepwise linear regression	Linear regression	Theoretical

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 Co-operative 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:

ER	=	erosion rate (µm/year)
P	=	precipitation rate (m/year)
SO_2	=	sulphur dioxide concentration (µg/m^3)
O_3	=	ozone concentration (µg/m^3)
H^+	=	acidity (meq/m^2/year)
R_H	=	average relative humidity, %
f_1	=	$1\text{-}\exp[-0.121.R_H/(100\text{-}R_H)]$
f_2	=	fraction of time relative humidity exceeds 85%
f_3	=	fraction of time relative humidity exceeds 80%
TOW	=	fraction of time relative humidity exceeds 80% and temperature >0°C
ML	=	mass loss (g/m^2) after 4 years
MI	=	mass increase (g/m^2) after 4 years
CD	=	spread of damage from cut after 4 years, mm/year
Cl^-	=	chloride deposition rate in mg/m^2/day
$Cl_{(p)}^-$	=	chloride concentration in precipitation (mg/l)
D	=	dust concentration in mg/m^2/day

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:

$$P \cdot H^+ \text{ (mg/l)} = 0.001 \cdot H^+ \text{ (acidity in meq/m}^2\text{/year)}$$

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.

15.4.1 Natural Stone

The types of stone commonly used for buildings and monuments include granite, sandstone, limestone, marble and slate. The durability of these stones is 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.

The full impact pathway for stone was shown in Figure 15.1. 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 Gibbs, 1994; Short, 1994).

The weathering of stone occurs naturally, primarily because of carbon dioxide present in the atmosphere. CO_2 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 CO_2 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 SO_2 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 SO_2 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 SO_2 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 SO_2 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 NO_x may also enhance these reactions involving SO_2.

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 CO_2, (ii) acceleration owing to rain acidity as a result of air pollution, (iii) attack by dry deposition of gaseous pollutants especially SO_2.
2. **Stage II**

The function (equation 15.1) 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 [15.2].

Butlin - Portland limestone:

$$ER = 2.56 + 5.1 \cdot P + 0.32 \cdot SO_2 + 0.083 \cdot H^+ \qquad [15.2]$$

The most recent dose-response functions for limestone samples are obtained from the four year exposures within the International Co-operative 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 [15.3].

ICP - unsheltered limestone (4 years):

$$ML = 8.6 + 1.49 \cdot TOW \cdot SO_2 + 0.097 \cdot H^+ \qquad [15.3]$$

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 [15.4] 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 [15.5], showing average mass loss over the total four year exposure.

Butlin - sandstone:

$$ER = 11.8 + 1.3 \cdot P + 0.54 \cdot SO_2 + 0.13 \cdot H^+ - 0.29 \cdot NO_2 \qquad [15.4]$$

ICP - unsheltered sandstone (4 years):

$$ML = 7.3 + 1.56 \cdot TOW \cdot SO_2 + 0.12 \cdot H^+ \qquad [15.5]$$

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 15.6 and 15.7) do not quantify later damages, such as exfoliation.

ICP - sheltered limestone (4 years):

$$MI = 0.59 + 0.20 \cdot TOW \cdot SO_2 \qquad [15.6]$$

ICP - sheltered sandstone (4 years):

$$MI = 0.71 + 0.22 \cdot TOW \cdot SO_2 \qquad [15.7]$$

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 [15.6] and [15.7] 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.

The ExternE assessment has relied on the functions 15.3 and 15.5 because of the duration of exposure and the pan-European nature of the work.

15.4.2 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 that 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) was 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 suggests 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, 1994) and therefore equations [15.4] and [15.5] 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, 1994). It should be noted that this problem may be more important in very old masonry.

15.4.3 Concrete

Portland cement, the major binding agent in most concrete, is an alkaline material that is susceptible to acid attack. Potential impacts to concrete include soiling/discoloration, 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.

15.4.4 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 15.3. 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 factors are 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 [15.8], 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:

Haynie - carbonate paint:

$$\Delta ER/t_c = 0.01 \cdot P \cdot 8.7 \cdot (10^{-pH} - 10^{-5.2}) + 0.006 \cdot SO_2 \cdot f_1 \quad [15.8]$$

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 15.9) is also considered to provide a range for the assessment.

Haynie - silicate paint:

$$\Delta ER/t_c = 0.01 \cdot P \cdot 1.35 \cdot (10^{-pH} - 10^{-5.2}) + 0.00097 \cdot SO_2 \cdot f_1 \quad [15.9]$$

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 [15.10], 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.

ICP - paint (4 years):

$$CD = -6.1 + 0.18 \cdot SO_2 + 0.18 \cdot O_3 \qquad [15.10]$$

15.4.5 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.

15.4.6 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, 1990) 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 [15.11], 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):

$$ML = [t^{0.78} + 0.46\log_e(H^+)]\cdot[4.24+0.55\cdot f_2\cdot SO_2+0.029\cdot Cl^-+0.029\cdot H^+] \qquad [15.11]$$

The second function we have considered is from Butlin *et al* (1992a), shown in equation [15.12], 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):

$$ER = 1.38 + 0.038\cdot SO_2\cdot + 0.48P \qquad [15.12]$$

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 [15.13] and [15.14], which have been the functions recommended under ExternE.

ICP - unsheltered zinc (4 years):

$$ML = 14.5 + 0.043\cdot TOW\cdot SO_2\cdot O_3 + 0.08\cdot H^+ \qquad [15.13]$$

ICP - sheltered zinc (4 years):

$$ML = 5.5 + 0.013\cdot TOW\cdot SO_2\cdot O_3 \qquad [15.14]$$

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.

15.4.7 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 [15.15]. The ICP study has also produced functions for aluminium samples after 4 years, shown in equations [15.16] and [15.17]. These could be recommended for areas where pollution levels are high, though for most areas damage to aluminium from air pollution is no problem.

Lipfert - aluminium (annual loss):

$$ML = 0.2 \cdot t^{0.99} \cdot (0.14 \cdot f_3 \cdot SO_2 + 0.093 \cdot Cl^- + 0.0045 \cdot H^+ - 0.0013 \cdot D)^{0.88} \qquad [15.15]$$

ICP - unsheltered aluminium (4 year):

$$ML = 0.85 + 0.0028 \cdot TOW \cdot SO_2 \cdot O_3 \qquad [15.16]$$

ICP - sheltered aluminium (4 year):

$$ML = -0.03 + 0.0053 \cdot TOW \cdot SO_2 \cdot O_3 + 74 \cdot Cl_{(p)}^- \qquad [15.17]$$

15.5 Estimation of Impacts on Materials

15.5.1 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 15.5.

Table 15.5. Averages of country-specific critical thickness losses for maintenance or repair measures assumed in the analysis

Material	Critical thickness loss
Natural stone	4 mm
Rendering	4 mm
Mortar	4 mm
Zinc	50 µm
Galvanised steel	50 µm
Paint	50 µm

For natural stone and mortar, it is assumed that maintenance action will be required after 4 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.

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.

15.5.2 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 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 for the repair costs have been taken from different sources. For the UK estimated repair costs are taken from unit cost factors for each of the materials for which assessment was performed. These figures are based on data from ECOTEC (1996) and Lipfert (1987). For Germany repair costs have been obtained from inquiries with German manufacturers. Finally, damage costs given in a study for Stockholm, Prague and Sarpsborg (Kucera *et al*, 1993b) are also considered. Table 15.6 summarises the damage costs used in this analysis in 1995ECU.

Table 15.6 Repair and maintenance costs [ECU/m^2] applied in analysis

Material	**ECU/m^2**
Zinc	25
Galvanised steel	30
Natural stone	280.3(at)-4.8(ed)9 Tc0r(at)-a57c0.0025 Tw[(7c

15.5.3 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 SO_2 and NO_x 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 5.1). 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 around 300 ECU/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, 1995b) 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.

15.6 Uncertainties

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 O_3 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 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;
- Macroeconomic effects.

15.7 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.

15.8 References

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16. QUANTIFICATION OF OZONE DAMAGES

16.1 Introduction

Tropospheric ozone causes significant harm to human health and to agriculture. To guide environmental policy on emissions of the ozone precursors, NO_x and VOC (volatile organic compounds), one needs estimates of the damage per tonne of precursor. The calculation of damage costs involves the following steps (EC 1995b):
i) calculate the incremental ozone concentration, per precursor emission from a given source;
ii) calculate the physical damage for each receptor (humans, crops, etc.) that may be affected;
iii) multiply by the cost per damage;
iv) sum over all regions where damage occurs.

Site dependence should be taken into account, especially in the formation of ozone (highly dependent on climate) and the distribution of receptors relative to sources.

A great deal of work has been and is being done on various aspects of this problem, but calculations of damage costs have rarely been carried out; in fact, only in recent years have the elements of such a calculation been sufficiently well established to permit meaningful estimates. In the USA ozone damage estimates have recently been published as part of an evaluation of the external cost of electricity (ORNL/RFF 1994, Rowe et al 1995). In Europe an analogous evaluation has been done for a power plant in Germany (EC 1995c), but the ozone modelling was restricted to the local range, up to about 75 km from the source. Our

As for the geographical scale, the analysis needs to extend far enough to capture all significant contributions. Even if the incremental concentration is very small at large distances, the contribution to the total damage can be appreciable because the number of receptors is also large. For pollutants whose residence time in the atmosphere is on the order of days, such as particulates, NO_x and SO_x, the range of the damage calculation needs to be extended to thousands of km if the exposure-response function is linear (see Chapter 4). For ozone the range is even larger because certain precursors have residence times of months (CO) or years (CH_4).

In the present paper we combine the results of the ozone models of EMEP and Harwell with the exposure-response functions and economic valuation for health and agricultural crops recommended in this report. Due to the limited geographical resolution of the available ozone modelling data, our estimates are only for average emissions in Europe (EMEP) or at 52°N latitude (Harwell). The nature of this work necessitates drastic approximations to compensate the lack of data, and we caution therefore that current estimates are no better than an order of magnitude.

We do not include effects of ozone on global warming. Even though they appear to be sizeable, they are far less troubling than the long-lived greenhouse gases (CO_2, CH_4, N_2O, ...): most of the tropospheric ozone would disappear within weeks after a reduction of the precursors.

16.2 Ozone Formation

From major reviews (e.g. PORG 1990, PORG 1993, Seinfeld 1986, Zannetti 1990), it is clear that ozone differs from other major air pollutants in important ways:

- its formation is dependent on reactions involving two principal classes of pollutants, NO_x and VOC;
- the formation of O_3 is complex and non-linear, involving hundreds of chemical reactions;
- because some of the reactions are photolytic, concentrations are critically dependent upon meteorology, and ozone formation varies geographically, diurnally and seasonally;
- some of the impacts (e.g. on crops) may have non-linear exposure-response functions and thresholds, so that the impacts may not be a simple function of annual cumulative dose.

From more detailed analyses, the following simplified theory may be deduced. 90% of atmospheric ozone is in the stratosphere where it is produced in reactions involving ultra-violet radiation. Ozone is also present in smaller concentrations as a natural component of the troposphere. This is due largely to periodic invasions of stratospheric air across the tropopause, but also reactions involving a range of trace constituents.

The main sequence of reactions which creates ozone in the free troposphere is through the photolytic destruction of NO_2:

$$NO_2 \xrightarrow{J} NO^+ \tag{16.1}$$

and

$$O_2 + O \rightarrow O_3 \tag{16.2}$$

This is counterbalanced by the ozone destroying reaction:

$$NO + O_3 \xrightarrow{K} NO_2 + O_2 \tag{16.3}$$

In the absence of light, this last reaction is dominant, and ozone levels are reduced at night where excess NO is present. In the daytime, the reactions may reach equilibrium so that the ozone concentration is given by:

$$[O_3] = \frac{J[NO_2]}{K[NO]} \tag{16.4}$$

in other words, ozone levels are determined by the ratio of the concentrations of NO and NO_2.

Particularly in the polluted boundary layer, this ratio can be significantly affected by the presence of VOC. The process involves a series of complex reactions, involving a range of hydrocarbon radicals, R. In essence, the formation of NO in Equation 16.3 is supplemented by reactions, involving the alkyl peroxy radicals, RO_2, which can be described as follows:

$$RO_2 + NO \rightarrow RO + NO_2 \tag{16.5}$$

These reactions oxidise NO to NO_2, without reducing ozone, and therefore increase the net rate of formation of ozone significantly.

The reactions which form alkyl peroxy radicals occur most rapidly in the presence of high levels of sunlight, NO_x and VOC (PORG, 1990). High ozone concentrations at ground level are characteristic of stable atmospheric conditions in summer, when ozone is produced rapidly and not dispersed.

Both observations and model predictions indicate that mean tropospheric ozone levels have doubled over the last century (Houghton et al, 1996). The pollutants and processes responsible are currently the subject of major research exercises, primarily in the context of studies of tropospheric ozone as a greenhouse gas. Tropospheric models indicate that a range of pollutants have a role - notably NO_x, carbon monoxide, methane and other VOC.

The category of VOC lumps together substances whose ozone creation potential is quite different (Derwent, Jenkin and Saunders 1996). Methane is often considered in a category apart: because its residence time in the atmosphere is much longer than the other VOC its contribution to ozone formation is significant only at the global, not at the local or regional level.

With the exception of methane, VOC have relatively short lifetimes and therefore are expected to play a reduced role in the chemistry of the free troposphere compared to the boundary layer.

The relative importance of different substances and processes seems to be different between the boundary layer and the free troposphere; in the latter, longer lived species such as carbon monoxide are likely to be more important. Here NO_x is present at very low concentrations, but nevertheless plays a key role in ozone formation. The dominant ozone producing reactions are similar to those in the boundary layer, except that methane is the only significant VOC involved. Methane is also an important sink for ozone. At very low NO_x concentrations, methane has a net ozone reducing role. But at NO concentrations in excess of 20 ppt, the ozone creation role of the methyl peroxy radical (Equation 16.5 above) is more important (Houghton et al, 1992).

It is clear that NO_x plays an important role in observed ozone increases in the free troposphere although quantification remains uncertain. Preliminary numerical estimates of the relative importance of different ozone precursors were presented by the IPCC (Houghton et al, 1990), but later replaced by qualitative assessments (Houghton et al, 1992, Houghton et al, 1996). These indicate that NO_x is the most important of all the species involved, even at the global scale.

16.3 Incremental O_3 from Incremental Precursor Emissions

16.3.1 Models and Metrics

In view of the differences between boundary layer and free troposphere we use two ozone models to link incremental O_3 to incremental precursor emissions:

1. Europe with the EMEP model (Eliassen and Saltbones 1983, Simpson 1992), which covers all of a resolution of 150 x 150 km, and which focuses on boundary layer processes;
2. the Harwell Global Ozone model (Hough 1989 and 1991), which covers the entire earth with a latitudinal resolution indicated in Table 16.3 without distinction of longitudes, and which includes the free troposphere.

Because of the different nature of the data we have from these two models, the damage calculations are not directly comparable. The overlap between EMEP and the European portion of the Harwell model is not clear, and the emissions scenarios are different: European averages of NO_x and of VOC for EMEP, emissions of NO_x only at 52°N for Harwell.

For estimates of total damage the sensitivity to local detail is probably less than for primary air pollutants, because peak O_3 concentrations occur at tens of km from the sources of NO_x and VOC. These primary pollutant sources tend to be correlated with population, because motor vehicles are the dominant source of both NO_x and VOC.

There is a reduction of ozone near the emission source, since most NO_x is emitted in the form of NO, an ozone scavenger; ozone production occurs after the oxidation of NO to NO_2. Thus the correlation between ozone concentration and population density is reduced compared to primary pollutants.

One of the problems encountered in this work is the profusion of different indices that have been used for reporting ozone concentrations. In addition to 24 hr averages, people have used daily maxima during intervals such as 1 hr, 6 hr, 8 hr and daylight hours. Furthermore, each of these may be reported as averages over different periods, for instance the year or the summer season (with various lengths). Often the concentrations are reported as AOTx (Accumulated Ozone exposure above Threshold x, in ppb·hr) above a specified threshold, common choices being x = 30 ppb, 40 ppb, and 75 ppb. We refer to these indices as "ozone metrics".

The rationale for these choices is the desire to account for threshold effects (see e.g. Lefohn et al 1987). Unfortunately there is no obvious best choice; rather, it is likely that different end points have quite different thresholds. Even worse, the damage may vary with the time sequence of the exposure, for the same total exposure (see e.g. Nussbaum et al 1995). For an exact calculation of damages, one would need hourly ozone data as well as exposure-response relationships with a level of detail that will not be available in the foreseeable future.

In practice one takes the shortcut of using common ozone metrics to characterise both the output of an ozone model and the exposure-response relationships. Unfortunately the metrics that we have for the above ozone models are different from each other and from the metrics used by the available exposure-response functions. Naturally one looks for conversion factors between the metrics. However, such conversion factors vary with site and season; a single factor is necessarily an approximation. As an illustration, and without any attempt at being systematic we show a few conversion factors in Table 16.1. In view of the climate dependence we have put more weight on data from Europe than from the USA.

16.3.2 The EMEP Model

Simpson (1992 and 1993) has calculated, with 6 hr time resolution, the daily peak concentrations of O_3 during the six months April - September, for 1985 and 1989 and for four scenarios of precursor emission reductions. Simpson's calculations are based on annual emissions data of NO_x and VOC in each of the countries of Europe, including the European part of the former Soviet Union. The NO_x is specified as equivalent mass of NO_x, but it is assumed emitted as NO, taking into account ozone scavenging near the source (Simpson personal communication). In the following we use his results for the six month averages of the daily peak concentrations ("mean of daily maximum ozone concentrations" in Simpson's papers).

Table 16.1. A sample of conversion factors for ozone metrics, and choice for present paper.

a) ratio of concentration values 1-hr max. to 8-hr max

Amsterdam [a]	1.14
Rotterdam [a]	1.18
Paris, annual average [a, b]	1.58
Paris, summer [b]	1.5
Paris, winter [b]	2.0
London [a]	1.32

[a] data from APHEA project (Hurley personal communication)
[b] Dab et al (1995)

b) ratio of concentration values 1-hr max. to 24-hr

USA [c]	2.8

[c] Schwartz (1994, discussion p.653)

c) ratio of concentration values 8-hr max. to 24-hr

Germany (Baden Würtemberg) [d]	
Rural	1.0-1.2
Urban	1.3-1.9

[d] W. Krewitt (personal communication)

d) Choice for this paper (we neglect difference between 6-hr and 8-hr)

1-hr max. to 8-hr max	1.3
8-hr max. to 24-hr	1.5

In Table 16.2 we summarise Simpson's results for the four scenarios of precursor emissions. From the O_3 reductions in Table 16.2 one can calculate the reduction per tonne of NO_2 and per tonne of VOC at an average level, the average being over the summer season and over the countries of Europe. Due to the non-linearity's of ozone formation, the reductions per tonne of NO_2 are not the same at 100% VOC and at 50% VOC. Since the current emissions are close to those of 1985 and 1989, we take the 100% VOC scenario and an ozone reduction of 0.37 ppbO_3 per Mt/yr of NO_2. This is the average (seasonal and regional) change in 6hr peak O_3 concentration. The sensitivity to the other precursor, VOC, is almost as large, 0.31 ppbO_3 per Mt/yr of VOC.

These numbers are an aggregate at the European level. Table 8 of Simpson (1992) indicates significant geographic variability: the % O_3 reduction (for which we take a geographic average of 7%) for a 50% NO_x reduction ranges from a few percent (1% in Netherlands and 3% in UK) to a high well above ten percent (14% Switzerland, 13% Italy and Spain). A North-South gradient is to be expected in view of the role of solar radiation.
If diurnal patterns do not change, the % reduction can also be applied to daily averages; of course the magnitude of the ppb changes would be different.

Table 16.2 Scenarios for precursor emissions (Mt/yr) and resulting reductions of O_3 concentrations in Europe, from Tables 1 and 6 of Simpson (1993). The ozone metric is the daily 6 hr peak concentration, averaged over the 6 months April - September and over all countries of Europe.

Emission scenario		Emission Mt/yr [a]		O_3 reduction		**ppbO_3 per MtNO_2/yr**	**ppbO_3 per MtVOC/yr**
NO_X	VOC	**NO_2**	**VOC**	% [b]	ppb [c]		
100%	100%	21.0	22.0	0	0.00		
50%	100%	10.5	22.0	7	3.92	**0.37**	
100%	50%	21.0	11.0	6	3.36		**0.31**
50%	50%	10.5	11.0	10	5.60	0.21	0.15

Notes: [a] emissions for 1985 and 1989, rounded from Table 1 of Simpson (1993); NO_X emissions are NO_2 equivalent

[b] from Table 6 of Simpson (1993)

[c] for 56 ppb = mean of the 28 countries in Table 8 of Simpson (1992); Figures 1 and 2 of Simpson (1993) imply that the mean is between 50 and 60 ppb.

16.3.3 The Harwell Global Ozone Model

Table 16.3 shows results from the Harwell global ozone model (EC 1995b) for the ozone increment in the Northern Hemisphere due to incremental emission of NO_x at a latitude of 52 degrees; they are stated in terms of N, not NO_2. The effects are in terms of ppb ozone increment in different latitude bands, i.e. there is no longitudinal variation, which is obviously a severe approximation. Southern hemisphere increments are assumed negligible because of the low rate of mixing between hemispheres.

Table 16.3. Incremental tropospheric ozone concentration from an incremental of 1 Mt of N emitted between 48.6 and 56.4°N. The ozone metric is the 24 hr average concentration.

	Ozone Concentration Increment, ppb/Mt N				
Latitude range	**January**	**April**	**July**	**October**	**Annual average**
S Hemisphere	0.000	0.000	0.000	0.000	0.000
0.0- 4.8	0.013	0.002	0.000	0.000	0.004
4.8- 9.6	0.023	0.006	0.001	0.001	0.008
9.6-14.5	0.036	0.014	0.005	0.006	0.015
14.5-19.5	0.055	0.030	0.010	0.012	0.027
19.5-24.6	0.081	0.056	0.021	0.025	0.046
24.6-30.0	0.111	0.097	0.038	0.044	0.073
30.0-35.7	0.149	0.158	0.073	0.079	0.115
35.7-41.8	0.109	0.264	0.152	0.139	0.166
41.8-48.6	-0.110	0.433	0.364	0.200	0.222
48.6-56.4	-0.266	0.480	0.917	-0.008	0.281
56.4-66.4	-0.194	0.479	0.409	0.089	0.196
66.4-90.0	-0.151	0.391	0.274	0.117	0.158

16.4 Health Impacts

16.4.1 Exposure-Response Functions and Economic Valuation

While there is a general consensus among public health experts that exposure to ozone is harmful, definite quantitative relationships are difficult to establish. Several exposure-response functions have been published in recent years for mortality and for morbidity end-points such as hospital admissions and asthma attacks. Typically they are presented as linear functions of 1 hr or 8 hr peak concentrations, i.e. without threshold. Note that they are at the level of the entire population, not of individuals. Because of differences between individuals it is quite possible for the effective exposure-response function of a population to look approximately linear even if there are thresholds for individuals - as appears to be the case for mortality from particulates (Dockery et al 1993).

As one possible set of exposure-response functions we use the ones recommended by the ExternE Program (EC 1995b) and shown below in Table 16.4; they are based on 1 hr peak concentrations. Because of linearity we have expressed them as a coefficient c_o, the number of incidents per person per ppb per yr. To convert from 1 hr peak to the 6 hr peak concentrations of EMEP, we have multiplied by a factor 1.3, as suggested by the average of Table 16.1a. We neglect the difference between 6 hr and 8 hr peaks.

For a confirmation of the exposure-response functions we cite an extensive series of studies in the Greater Paris area (ERPURS 1994, Dab et al 1995) which is continuing as part of the APHEA project of the EC. The ERPURS results were obtained with a time series Poisson regression analysis of a population of 6.1 million people from 1987 to 1992. Several endpoints are found to be correlated with ozone concentration. Some of the endpoints are not common to both studies (e.g. minor symptom days in EC (1995b) and house calls in ERPURS), but for those that are, the results are within a factor of two of the ones chosen here.

Table 16.4 Exposure-response function slope c_o for O_3 health impacts and the corresponding costs. c_o includes a factor 1.3 to convert from 1 hr peak to 6 hr peak.

	$\mathbf{c_o}$ (cases per person·ppb·yr) [a]	ECU per case [b]	ECU/(yr·person·ppb)	%
Acute mortality (Sunyer et al 1996)	1.17E-05	82500 [c]	0.965	36%
HA Respiratory (Ponce de Leon 1996)	1.42E-05	6600	0.093	3%
RADs (Ostro et al 89)	1.95E-02	62	1.209	45%
Symptoms Days (Krupnick et al 90)	6.60E-02	6.3	0.416	15%
Total ECU/(yr·person·ppb)			**2.68**	100%

RAD = Restricted Activity Days, HA = Hospital Admission

[a] coefficients of Table 4.4 of EC (1995b) and updates (this report, Chapter 8), multiplied by 1.3

[b] from Table 4.3 of EC (1995b), except for acute mortality, see text below

[c] assuming average life span reduction of 9 months and value of 1 year of life lost = 110 kECU

For the economic valuation the ground rule is, by general consensus, to follow individual preferences. Hence one should use the willingness-to-pay (WTP) to avoid illness or death, rather than cost of illness (COI) or human capital. Unfortunately there are hardly any WTP data for morbidity, and even COI data are quite uncertain. For morbidity endpoints we adopt the values that have been used in the ExternE Program (EC 1995b), shown in Table 16.4; they are based mostly on COI data from the USA. The transferability of these values outside the European Union is, of course, questionable, especially for countries with much lower GDP per capita.

For the reference value for the protection of life (often called value of statistical life VOSL) the ExternE Program has adopted 2.6 MECU ($ 3.2 million). This is well in the mainstream of current economic thinking as indicated by a survey (Ives, Kemp and Thieme 1993) of 78 VOSL studies published between 1973 and 1990 in Europe and North America, for which the median is £$_{1990}$ 1.49 million ($ 2.7 million). The geometric standard deviation of these 78 studies is 3.4, i.e. the 68% confidence interval extends from $ 0.8 to 9.2 million. Such large uncertainty is not surprising for a good whose valuation is so problematic and controversial. One of the principal methods, contingent valuation (CV), is fraught with risks of overestimation, as highlighted in a paper by Desaigues and Rabl (1995) who identify several biases and find a CV estimate of 5.5 MFF ($ 1.1 million) for France.

In the spirit of economic rationality we base the valuation of acute mortality on YOLL (years of life lost) rather than the number of premature deaths. The relation between the value of 1 YOLL and VOSL depends on the discount rate, 1 VOSL being supposed equivalent to a discounted series of annual YOLL values for a period that corresponds to a typical loss in VOSL studies. We adopt a value of 110 kECU for 1 YOLL (see Chapter 12).

One also needs to estimate the life span reduction for acute mortality. There are no data, but epidemiologists think that the reduction might be in the range of weeks to months, perhaps a few years (Hurley personal communication). Probably there is a wide range of values, and a lognormal distribution seems plausible. For YOLL valuation one needs the average of this distribution. For an illustration suppose the real distribution has median 2.5 months and geometric standard deviation $\sigma_G = 5$; then the average is 9 months, and the 1 σ_G interval (68% of cases) goes from 0.5 months to 12.4 months. The average is much higher than the median. For the ExternE project an average life span reduction of 9 months has been adopted as working hypothesis (see Chapter 8).

Thus the cost of acute mortality is uncertain for several reasons: the exposure-response function, the value of 1 YOLL, and the life span reduction. Our choice of 110 kECU per YOLL is too high for many countries outside the EU since WTP is roughly proportional to disposable income - although for reasons of equity one may prefer a single value, a common practice in this field that we shall follow. Such a choice is not too unreasonable for the present paper since most of the poorer countries are far from the sources of precursor emissions considered here.

When c_o is multiplied by the respective cost per incident we obtain an effective exposure-response function for the cost per ppb of O_3, shown in the penultimate column of Table 16.4. The total cost is 2.68 ECU/yr·person·ppb. It is dominated by mortality and RADs.

We close this section with two remarks about the risk of our damage estimates being too high:
1. the total health damages are not very sensitive to further reduction in the VOSL as the mortality damages are only 36% of the total; and
2. the morbidity numbers may be under-estimates as they do not include the full WTP; and
3. we have only included the end points that are considered the most certain; for instance EC (1995b) cites an exposure-response function for asthma attacks (Holguin et al, 1985) that would add another 0.8 ECU/(yr·person·ppb) to Table 16.4.

16.4.2 Health Impacts for EMEP

The EMEP model (Simpson 1992, 1993) uses constant percentage changes in precursor emissions over Europe and gives average European ozone changes. We multiply this output by the cost per person per ppb to give an estimate of the total European damage. For linear exposure-response functions and uniform population density the precise geographic distribution of ozone reductions does not matter, and a simple linear estimate as in Table 16.4 is sufficient. Uniformity may not be such a bad approximation in view of the distance between ozone formation and population centres, as mentioned above.

The total population of Europe is 724 million. Multiplying the population by the total damage cost per person per annual average ppb in Table 16.4 and by the change in O_3 per tonne/yr of NO_2 from Table 16.2 we obtain

$$724 \times 10^6 \text{ persons} \times 2.68 \frac{\text{ECU}}{\text{yr} \times \text{person} \times \text{ppb}} \times 0.37 \textbf{ Fehler!} = \mathbf{718\ ECU/tNO_2} \tag{16.6}$$

For VOC we use the number 0.31 ppb/(Mt/yr VOC) from Table 16.2 to obtain **602 ECU/t VOC**. These are averages for emissions in Europe, and only damages in Europe are counted.

16.4.3 Health Impacts for Global Model

We have applied the concentration increments of the Harwell model to a world population database transformed from a national basis to latitude bands by inspection of maps; the population per latitude band is s

$$c_o(24hr) = 1.5 \times c_o(8hr) \tag{16.7}$$

Table 16.3 also needs to be multiplied by 14/46 to convert from t N to t NO_2. For example, the global damage in the 56.4-66.4°N latitude band is obtained as

$$80 \times 10^6 \text{ persons} \times 1.5 \times 2.68 \; \frac{\text{ECU}}{\text{yr} \times \text{person} \times \text{ppb}} \times 0.196 \times \frac{14}{26} \; \frac{ppb}{MtNO_2 yr} = 19.2 \; \frac{ECU}{tNO_2}$$

The advantage of this approach is that European and non-European impacts can be assessed separately. It is reassuring that the total health damage estimates for Europe are not too dissimilar between EMEP and Harwell, given the uncertainties and the model differences. Also of interest is the result that the global damage from the Harwell model is much larger than the damage in Europe alone, although this result may be exaggerated by a calculation that does not distinguish receptors close to and far from the source.

16.5 Agricultural Impacts

16.5.1 Exposure-Response Functions and Economic Valuation

A large number of laboratory experiments have clearly established that ozone, at concentrations commonly found in urban environments, has harmful effects on many plants. Exposure-response functions have been derived for several plants of economic importance. Nonetheless the quantification of crop damages is problematic. Even if we had complete, regionally disaggregated data bases for agricultural production, the available exposure-response functions are very incomplete: for many important crops we have little or no ozone information. Some functions are stated in terms of AOTx and we cannot use them because we do not have conversion factors to the EMEP and Harwell data.

Table 16.5 Distribution of population in the latitude bands of Harwell global ozone model, and the resulting damage cost per t/NO_2.

	Population		ECU/t NO_2	
Latitude	**Global**	**Europe**	Global	Europe
SH	603	0	0	0
0-4.8	132	0	1	0
4.8-9.6	254	0	2	0
9.6-14.5	363	0	7	0
14.5-19.5	344	0	11	0
19.5-24.6	580	0	33	0
24.6-30	623	0	56	0
30-35.7	757	1	107	0
35.7-41.8	590	65	120	13
41.8-48.6	538	216	146	59
48.6-56.4	398	361	137	124
56.4-66.4	80	72	19	17
66.4-90	10	9	2	2
Total	**5272**	**724**	**641**	**215**
%	100%	14%	100%	34%

Furthermore, laboratory experiments are typically carried out under very limited conditions (single species, single pollutant, particular exposure scenarios, controlled climate, ...), and one wonders to what extent they are representative of real growing conditions in a variety of countries and climates. As an example of possible complexities we cite a study by Nussbaum et al (1995) who subjected a mixture of perennial rye grass and white clover to several different ozone exposure patterns in the typical open-top chamber arrangement. This combination of plants was chosen because of their importance for managed pastures in Europe. The authors find that the ozone damage depends not only on the total exposure but also on the exposure pattern. Furthermore they see two thresholds: species composition is fairly well correlated with AOT40 but total forage yield with AOT110.

It is interesting to compare exposure-response functions for different species. Exposure-response functions for several crops are presented elsewhere in this report. They give the crop yield y as function of O_3 concentration, y being normalised to unity at Conc. = 0. They were derived from experiments in the USA and their functional form is Weibull. In the present note we are concerned with marginal changes around current concentration values. Thus we consider the reduction in crop yield

$$\text{reduction in yield per ppb} = \frac{1}{y}\ \frac{dy}{dConc} \qquad (16.8)$$

relative to current agricultural production. Since the functions in Table 8.3 of EC (1995b) refer to peak concentrations during 7 or 12 hr periods, we use the 6 hr peak values of Simpson directly. In Figure 16.1 we plot the yield reduction for several published exposure-response functions.

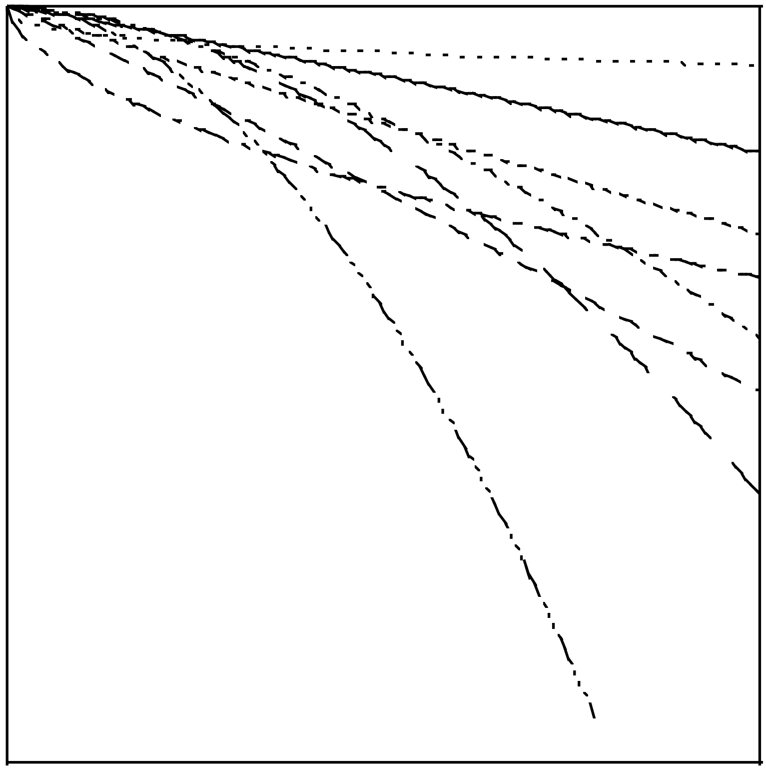
3

Table 16.7 below. A major agricultural product is not included in Table 16.7: forage for production of cattle. But this does not seem important since the main ingredient of forage, rye grass, is quite insensitive to ozone.

16.5.2 Agricultural Impacts for EMEP

To evaluate the crop losses we use the data for agricultural production 1989 in the EU, according to Table 8.1 of the Methodology Report EC (1995b). They are reproduced here in Table 16.7. For those crops whose exposure-response functions are known we evaluate the yield reduction (3rd column) with the respective function; for the others we use the average of Table 16.6, rounded to -0.006. The sum of the crop losses is 516 MECU/ppb.

To obtain the damage per tonne of NO_2, we multiply by the change in O_3 per tonne/yr of NO_2 from Table 16.1 as above

$$516.2 \text{ MECU/yr per ppb} \times 0.37 \text{ ppb/MtNO}_2 = 193 \text{ ECU/tNO}_2 \cong \mathbf{200\ ECU/tNO_2}.$$

For VOC the analogous result is

$$516.2 \text{ MECU/yr per ppb} \times 0.31 \text{ ppb/MtNO}_2 = \mathbf{160\ ECU/tVOC}.$$

Table 16.7 Cost of crop losses.
2nd column = agricultural production 1989 in EU, 3rd column = yield reduction,
4th column = product of 2nd and 3rd column.

	10^9 ECU/yr [a]	Yield reduction [b]	MECU/ppb
Fresh vegetables	17.5	-0.0048	-83.2
Wheat and spelt	12.5	-0.0047	-58.9
Other crops	12.1	-0.0058	-69.9
Grape and wine	11.7	-0.0058	-67.6
Fresh fruit	9.2	-0.0058	-53.1
Barley	4.7	-0.0058	-27.1
Sugar beet	4.7	-0.0115	-54.2
Oil seeds and fruit	4.5	-0.0058	-26.0
Potatoes	4.4	-0.0058	-25.4
Maize	3.9	-0.0053	-20.8
Olive oil	2.9	-0.0058	-16.7
Citrus fruit	2.3	-0.0058	-13.3
Total	**90.4**		**-516.2**

[a] from Table 8.1 of Methodology Report EC (1995b)

[b] yield reduction at 56 ppb: species specific value from Figure 1 where available, otherwise average according to Table 16.6.

16.6 Agricultural Impacts of NO_2 for Global Model

Calculation of global scale damages for crops is problematic. Geographically disaggregated databases of crops are not available. The crop analysis is therefore not based on accurate data on crop yields. Reliable exposure-response functions are also unavailable for many crops, and even where they are available, they are generally non-linear so that an accurate calculation of marginal damages requires an assessment of baseline ozone concentrations, for which global databases are not available. As non-European losses are less than European losses and crops' damages are less than health damages, an approximate technique is adequate.

The global crop inventory is assessed using a simple relationship between the fraction, F, of GDP which is provided by agriculture and per capita GNP, G, in US$. This relationship describes the distribution of agricultural production between countries fairly well (Downing et al 1996). It is as follows:

$$\log_e F = 2.1 - 0.55(\log_e G) \qquad (16.9)$$

It is assumed that only arable production is significantly sensitive to ozone. Agricultural GDP is therefore scaled by a factor of 48.7% taken from European data (EC 1995b). As for health impacts, for each country divided it into latitude bands (with pro rata population estimates), and the above linearised exposure-response function is used for all arable crops - a yield loss of 0.58% per ppb. The calculations are done as for health, except that the summer value for ppb ozone per Mt NO_2 from the Harwell global ozone model is used (corrected to an 8-hr metric), not the annual average. Results for ozone damages are shown in Table 16.8.

Table 16.8. Cost of crop losses for Harwell Global model.

European	340 ECU/t NO_2
Non-European	150 ECU/t NO_2
Global	490 ECU/t NO_2

16.7 Global Ozone Impacts due to VOC, CO and CH_4

Since the Harwell Global Ozone model gives us only the effects of NO_x, we estimate the global effects of the other ozone precursors on the basis of results provided by IPCC (Houghton et al 1990). The best measures of the relative impacts of carbon monoxide, methane and NMVOC (non-methane VOC) on global scale ozone production are estimates of the tropospheric ozone related components of (mass based) global warming potentials in the climate forcing literature. These are shown in Table 16.9.

Table 16.9. Global scale tropospheric ozone production potential, based on Houghton et al (1990).

	relative to CO_2	relative to NO_x
NO_x	150	1
CO	5	0.033
Methane	24	0.16
NMVOC	28	0.187

These factors are used here to estimate the relative non-European impacts of the non-NO_x ozone precursors. This is not a very reliable approach for a number of reasons:

- the non-methane GWP values of the 1990 IPCC report were not repeated in subsequent reports and the latest IPCC assessment concludes that GWPs for NO_x, VOC and carbon monoxide cannot be estimated accurately (Houghton et al, 1996)
- the lifetimes of the species are different and therefore the importance of the regional range contribution (which ought to be subtracted from the global value to give the non-European range value) will not be the same in each case
- the GWPs are based on radiative forcing which is indicative of concentration in the upper troposphere. Although there are reasonable theoretical and empirical reasons to believe that the ground level increment will be related to the increment higher in the troposphere, there is no reason to expect a very strong correlation.

Neglecting these complexities, we take the NO_x damage outside Europe as calculated above and multiply it by the ozone production potential relative to NO_x of Table 16.9 to estimate the damage of the other precursors. The results

16.8 Ozone Impacts on Materials

Ozone causes damage to some materials, in particular rubber products and surface coatings. A recent survey by Lee et al (1995) highlights the dearth of quantitative information on this issue. The only damage cost estimates in the literature stem from the USA around 1970, and Lee et al extrapolate them to the UK in 1995 to obtain an indication of the order of magnitude: £170 to 345 million/yr for an ambient 24-hr average concentration of 15 to 30 ppb. Lee et al also derive a direct estimate of damage to paint in the UK by using dose-response functions and inventories of painted surfaces; this yields a range from £0 to 12 million/ppb per year, compatible with the extrapolation from the USA. As a third significant source of information in the summary of Lee at al (their Table 6) one finds the cost of anti-ozonants that are added to rubber products. This number is £25 million/yr and amounts to about £1 million/ppb per year if one assumes linearity in ozone concentration (in citing this number we count only the direct costs of anti-ozonants to producers because the multipliers for indirect costs involve redistribution rather than new costs). Finally a fourth item in Table 6 of Lee et al, direct damage to tires, appears negligible.

Based on these numbers we take linearised ozone damage estimate in the range of £1.2 to 12 million/ppb per year or 1.5 to 15 MECU/ppb per year, for a population of 58 million. For comparison with the 6-hr metric used for European health damages in Table 16.4, we divide by 1.5 with the result of 1 to 10 MECU/ppb for 58 million or 0.17 ECU/ppb·person·yr. This is 0.4 % to 4 % of the magnitude of our European health cost estimate in Table 16.4. Scaled to the population of Europe it amounts to 12.5 to 125 MECU/ppb·yr, which is 2.4 to 24 % of the crop losses in Table 16.7. We conclude that ozone damage to materials is relatively small but potentially significant. Because of the small size and extreme uncertainty we have not included it in the summary below.

16.9 Summary and Uncertainties

We have estimated the damage cost per tonne of NO_2, for health impacts and crop losses. The results are summarised, after rounding, in Table 16.11. Since Simpson's numbers imply that the ppbO_3 per tonne of precursor for VOC and for NO_2 is in the ratio 0.31 to 0.37, the damage per tonne of VOC is about 16% lower than that per tonne of NO_2. For the Harwell model we have no information on damage per tonne of VOC.

Table 16.11. Summary of damage costs per tonne of NO_2.
a) Damages in Europe, based on EMEP for average emissions in Europe. Damage per t VOC is a factor 0.84 smaller.

	ECU/tNO_2	**\$/tNO_2**
mortality	259	324
morbidity	460	575
Crop losses	200	250
Total	**919**	**1149**

b) Damages, based on Harwell model for emissions at 52°N. Health impacts are based on annual averages, crop losses on July values in Table 16.3.

	ECU/tNO_2			**\$/tNO_2**
	Europe	Outside Europe	Total	Total
mortality	77	153	230	288
morbidity	138	272	410	513
Crop losses	340	150	490	613
Total	**555**	**575**	**1130**	**1414**

The close agreement between the totals for EMEP and Harwell is a coincidence. The geographic range is different. For the area common to both the difference is a factor of two (for crops) to four (for health). That the ratios differ between crops and health is consequence of different seasonal patterns (Apr.-Sept. for EMEP, July for Harwell crops, year round for Harwell health). The dose-response functions are the same throughout, and so is the valuation of health costs; for crops there is a difference in the economic valuation but it is insignificant compared to the difference in atmospheric modelling.

The two ozone models are very different as explained in Section 3. EMEP has a better geographic resolution and models only the boundary layer, while Harwell puts more weight on long distance transport and contributions from the free troposphere; the overlap between the models is not clear. Our main conclusion from the Harwell model is that impacts outside Europe appear be important. Combining the total damage from EMEP and the Outside Europe damage from Harwell, we find

$$919 + 575 = 1494 \text{ ECU/t } NO_2 \, . \qquad (16.10)$$

For VOC damage we distinguish methane and NMVOC. Ozone formation by methane is insignificant at the regional scale, and the EMEP results are essentially due to NMVOC. Hence we take NMVOC damage as 0.84 times the NO_2 damage from Table 16.11 and we add 28 + 130 ECU/t from the last line of Table 16.10 for global damage to obtain

$$0.84 \times 919 + (28 + 130) = 930 \text{ ECU/t NMVOC} \, . \qquad (16.11)$$

For methane we take 24 + 110 = 134 ECU/t from Table 16.10. The global VOC estimates are very uncertain, but at least for NMVOC their contribution is relatively small.

The uncertainties are large, easily a factor of 2 to 4 for the exposure-response function and damage per ppb, in the sense of geometric standard deviation. The estimation of ppbO_3 per tonne of NO_2 might likewise be in error by a factor of 2 to 4. If we take the rule

$$[\ln(\sigma_{gtot})]^2 = [\ln(\sigma_{g1})] + [\ln(\sigma_{g2})]^2 \qquad (16.12)$$

for combining the geometric standard deviations for a product of two factors, with σ_{g1} _ σ_{g2} _ 3 we find that the result has a geometric standard deviation of about 5 for damages valued according to market prices, i.e. crops, whose price uncertainty is negligible. For health impacts this is compounded by the uncertainty of the economic valuation, especially the value of life. In view of the controversies and uncertainties surrounding the mortality costs, we note that in our estimate mortality accounts for roughly a quarter of the total.

Our results are summarised in Table 16.12. They are aggregate numbers at the European level. For the variation with emission site the numbers in Table 8 of Simpson (1992) suggest that the ppbO_3 per tonne of NO_2 may vary by about a factor of two around the mean between Northern and Southern Europe.

Table 16.12. Summary of damage costs per tonne of precursor for average emissions in Europe. The uncertainty is large, estimated to correspond to a geometric standard deviation of about 5 (multiplicative confidence interval).

	ECU/t	**$/t**
NO_2	1500	1875
NMVOC	930	1160
CH_4	130 ?	160 ?

It is interesting to compare with the estimate of local ozone damage made for Lauffen, Germany, on the basis of a short range ozone model calculation for a single peak ozone episode. Summing the ozone impacts in Table 2a, p.23 of (EC 1995c), we find 0.35 mECU/kWh for an emission rate of 0.8 gNO_2/kWh, implying 440 ECU/t NO_2 (550 $/t). We do not know to what extent local damages would be additive to our regional estimates. We also note that our results are consistent with a recent study of ozone impacts in Belgium (Mayeres and Proost 1996) although detailed comparison is not meaningful because there are too many differences; for instance the atmospheric model is for the Netherlands.

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17. NOISE AND AMENITY

17.1 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 that 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 was therefore 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 Pascal's) by the equation:

$$Noise\ level = 20.\log_{10} \, {}^{Sound\ pressure}\!\big/_{20\mu Pa} \qquad 17.1$$

By measuring decibels on a logarithmic scale to the base 10, the noise level at 60 dB is ten times as intense as at 50 dB (the psychological sensation of loudness roughly doubles with each 10 dB increase in the intensity level). Ear sensitivity is greatest in the middle range of frequencies and so weighting scales are available that selectively discriminate against sounds of very high and very low frequency. The most commonly used weighting scale is the A-scale denoted by dBA or dB(A). 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 that 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.

17.2 The Noise Impact Pathway

The noise impact pathway is shown in Figure 17.1. 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.

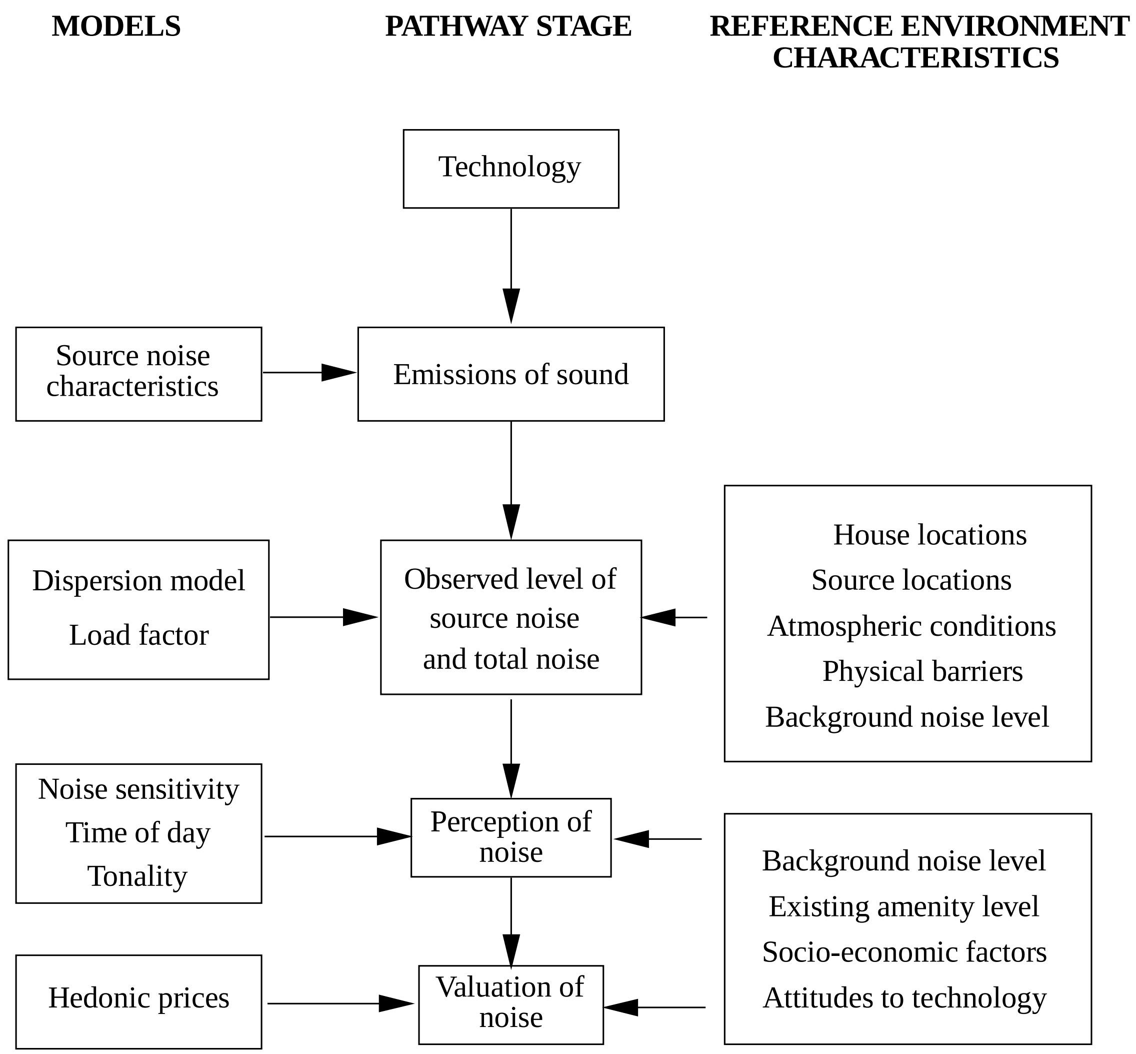

Figure 17.1 Impact pathway for noise.

17.3 Emissions of Sound

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.

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 between noise and emission level given in equation 17.2 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_{Aeq} measure that is of interest. For intermittent sounds it is the peak level.

17.4 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.

17.4.1 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:

$$\begin{aligned} L_p &= L_w - 10.\log_{10}(2\pi r^2) \\ &= L_w - 10.\log_{10}(2\pi) - 20.\log_{10} r \end{aligned} \qquad 17.2$$

This is the L_{Aeq} 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.log2 (approximately 6) dB(A) per distance doubling.

For cases where noise is believed to be a potentially important impact, consideration of more complex approaches is justified.

17.4.2 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 and Gustafsson, 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:

$$L_p = L_w - 10.\log_{10}(2\pi) - 20.\log_{10} r - \alpha r \qquad 17.3$$

where α 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, α 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 17.3 is:

$$L_p = L_w - 8 - 20.\log_{10} r - 0.05.r \qquad 17.4$$

17.4.3 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 17.4. 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.

17.4.4 Model Comparison and Choice

Comparison of the CONCAWE model (Manning, 1981) and the IEA model (Ljunggren and Gustafsson, 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 that are significant at these distances. At this range, the noise tends to attenuate in proportion to $10.\log_{10}r$, as opposed to $20.\log_{10}r$ 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.

17.5 Observed Sound Levels

The IEA model in the form of equation 17.4 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 that defines the dB, this requires use of the following formula:

$$L_{p,total} = 10.\log_{10} \sum_{i=1}^{N} 10^{L_{p,i}/10} \qquad 17.5$$

A similar approach is used to summing the source noise level, $L_{p,total}$, and the background noise level, L_{back}. The observed noise level is:

$$L_{obs} = 10.\log_{10}(10^{L_{p,tot}/10} + 10^{L_{back}/10}) \qquad 17.6$$

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).

17.6 Perception of the Observed Noise Levels

17.6.1 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 17.1 - 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 17.5). 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 17.1) 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 that have been developed in the general applications of noise analysis and control. Others are more difficult, as described below.

17.6.2 Time of Day Effects

The noise levels calculated by equations 17.3 to 17.6 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 that have been adopted in noise control represent a good measure of the average. The discussion in Section 17.1 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:

$$L_{dn} = 10.\log_{10} \; {}^{15}\!/\!_{24} 10^{L_{Aeq}/10} + {}^{9}\!/\!_{24} 10^{(L_{Aeq}+10)/10} \qquad 17.7$$

The L_{dn} levels for both the source and background noise are used as inputs to equation 17.6 to calculate the observed ‘day-night’ noise level, $L_{dn,obs}$.

17.6.3 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.

$$NNI = P\ (dB) + 15 \log n - 80 \qquad 17.8$$

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_{Aeq} 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, $L_{year,obs}$, is given by:

$$L_{year,\ obs} = 10.\log_{10} \left[f.10^{L_{dn,obs}/10} + (1 - f).10^{L_{dn,back}/10} \right] \qquad 17.9$$

The difference between this value, $L_{year,\ obs}$, and the expected noise without the turbines,
L

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 17.10 introduces a non-linearity (with respect to dB(A)) into the assessment, but allows analysis of noise on a probabilistic basis.

17.6.5 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.

17.6.6 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 Penrhyddlan and Llidiartywaun 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.

17.6.7 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 17.6), 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 17.1

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 17.8), as this values the incremental noise. In effect, therefore the 'dB(A) costing' method allows for a strong sensitivity to background noise.

Table 17.1 Effect of background noise on incremental observed noise.

Source noise, dB(A)	Background noise, dB(A)	Observed noise, dB(A)	Incremental noise, dB(A)
35	20	35.1	15.1
35	25	35.4	10.4
35	30	36.2	6.2
35	35	38.0	3.0
35	40	41.2	1.2
35	45	45.4	0.4
35	50	50.1	0.1

17.6.8 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.

17.7 Research on the Valuation of Noise Impacts

The main European studies for valuation of disamenity from noise are:
- The Roskill Commission (1970)
- Hoffman (1984)
- Pearce, Barde and Lambert (1984)
- Oosterhuis and Van der Pligt (1985)
- Larsen (1985)
- Pommerehene (1986)
- Pennington *et al* (1989)

The results of these studies are shown in Table 17.2. A more detailed discussion of them was given in our earlier report (European Commission, 1995a, b).

Table 17.2 Summary of European noise studies.

Noise Source	Noise Level	Author Method	Method	Depreciation in house price (or rent).
Road / Industrial	55 - 65 dB(A)	Oosterhuis & Van der Pligt (1985)	HPM	DFL 400, due to 1dB(A).
Road	30 dB(A)	Pommerehene (1986)		1% rent
	70 dB(A)		CVM	1.4% rent
Aircraft	NNI unit increase	Pommerehene (1986)	CVM	0.2% rent
Aircraft	NNI unit increase	Pennington, Topham & Ward (1989)	HPM	0.4 - 0.5% (6% for areas most affected).
Aircraft	NNI unit increase		HPM	Only slight (ACORN) negative noise effect
Aircraft	35 - 45 NNI	Roskill (1970)	HPM (H)	0 - 3.3% (G)4.5 - 16.4%
	45 - 55 NNI	Roskill (1970)	HPM (H)	(H)2.9 - 13.3% (G)10.3 - 29.0%
	55+ NNI	Roskill (1970)	HPM (H)	(H)5.0 - 22.5% (G)N/A
Aircraft	60 dB(A)+	Hoffman (1984)	HPM	1% per dB(A) increase.
Road	1000 ADT[1]	Larsen (1985)	HPM	0.8%

Notes:
ADT = average daily number of vehicles on the nearest road.
The Noise and Number Index (NNI) is defined as:
NNI = PNdB + 15 logn - 80
PNdB = log average of peak noise levels of aircraft heard.
n = number of aircraft (in the UK on a summer day).

Despite the non-inclusion of American studies in this report, it is still helpful to compare results. The NDSI (the percentage depreciation caused by a unit increase in the noise nuisance level) figure obtained by Pennington *et al* (1989); 0.4 - 0.5% (first result) is very similar to those figures obtained in earlier American works, and reported in surveys carried out by Walters (1975), Pearce (1978) and Nelson (1980). Indeed a small but statistically significant noise effect of 0.4% per unit of aircraft noise measure is the average value found in the American literature.

17.8 Method for Valuation of Noise Impacts

From the last section, a weighted average of European studies on noise values has been taken. The hedonic pricing method is used to estimate the depreciation in house prices as a function of ambient noise level, which defines a noise depreciation sensitivity index, NDSI. 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 17.7) and intermittency (equation 17.8), and the resulting value, $L_{year,obs}$, is used directly as an input to valuation.

The NDSI recommended above was based on studies considering road traffic noise. As noted in Section 17.6, 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 17.9). 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.

17.8.1 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:

$$AVN = \sum_{all\ positions} (L_{year,obs} - L_{dn,back}) \times N_{houses} \times A(P) \times NDSI\,, \qquad 17.11$$

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 that 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 17.6.1. 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.

17.8.2 Valuation by Annoyance Costing

The costing of annoyance is based on the use of equation 17.10. 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 17.10, it can be shown that P(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 that 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(HA) to hedonic price depreciation (Gildert, 1993).

Using equation 17.10 on the noise levels from the source, the P(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.

17.8.3 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 17.5. But in the annoyance costing approach, this effect is more than offset by the exponential form of P(HA) (equation 17.10). 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.

17.8.4 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 17.9, the annual value of noise, AVN, in this case is then given by:

$$AVN = \sum_{NNI > 30} [P + 15.\log n - 80] \times N_{houses} \times A(P) \times NDSI\,, \qquad 17.12$$

where N is the number of houses at that location and A(P) is the annuitised average house price.

17.9 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 air and of water, the spatial range of the problem is confined to the immediate environs of the emission source. 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 that 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.

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18. VISUAL AMENITY

18.1 Introduction

There are two types of externality resulting from fuel cycle activities that relate to visual amenity. First we have the costs associated with aesthetic intrusion for example from the presence of wind turbines in a natural landscape. Secondly there are the costs associated with reduced visibility resulting from emissions of certain pollutants into the atmosphere. Although both involve a change in visual amenity, they differ in the way that the costs are perceived by society, and are therefore discussed independently.

18.2 Visual Intrusion

The direct 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 intrusion 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 intrusion is a local scale impact. Because of the heterogeneous nature of landscape the visual effects of the same technology in different places can vary greatly. 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 WTP principle has been followed for valuation. It can be implemented in two 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 contingent valuation) 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.

18.2.1 Direct Valuation by Contingent Valuation

Direct application of CVM for valuation of amenity changes can only be carried out where CVM studies have been undertaken. The remit 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 (European Commission, 1995).

The individuals in the area around a new fuel cycle facility 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. 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 survey that asked 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 identified WTP for a wider package of changes. In assessing the aggregate damages of any scheme, it is not only necessary to measure individual WTP, but also to determine the extent of the affected population. This problem is addressed in more detail below.

18.2.2 Indirect Valuation

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 Chapter 17, 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 18.1.

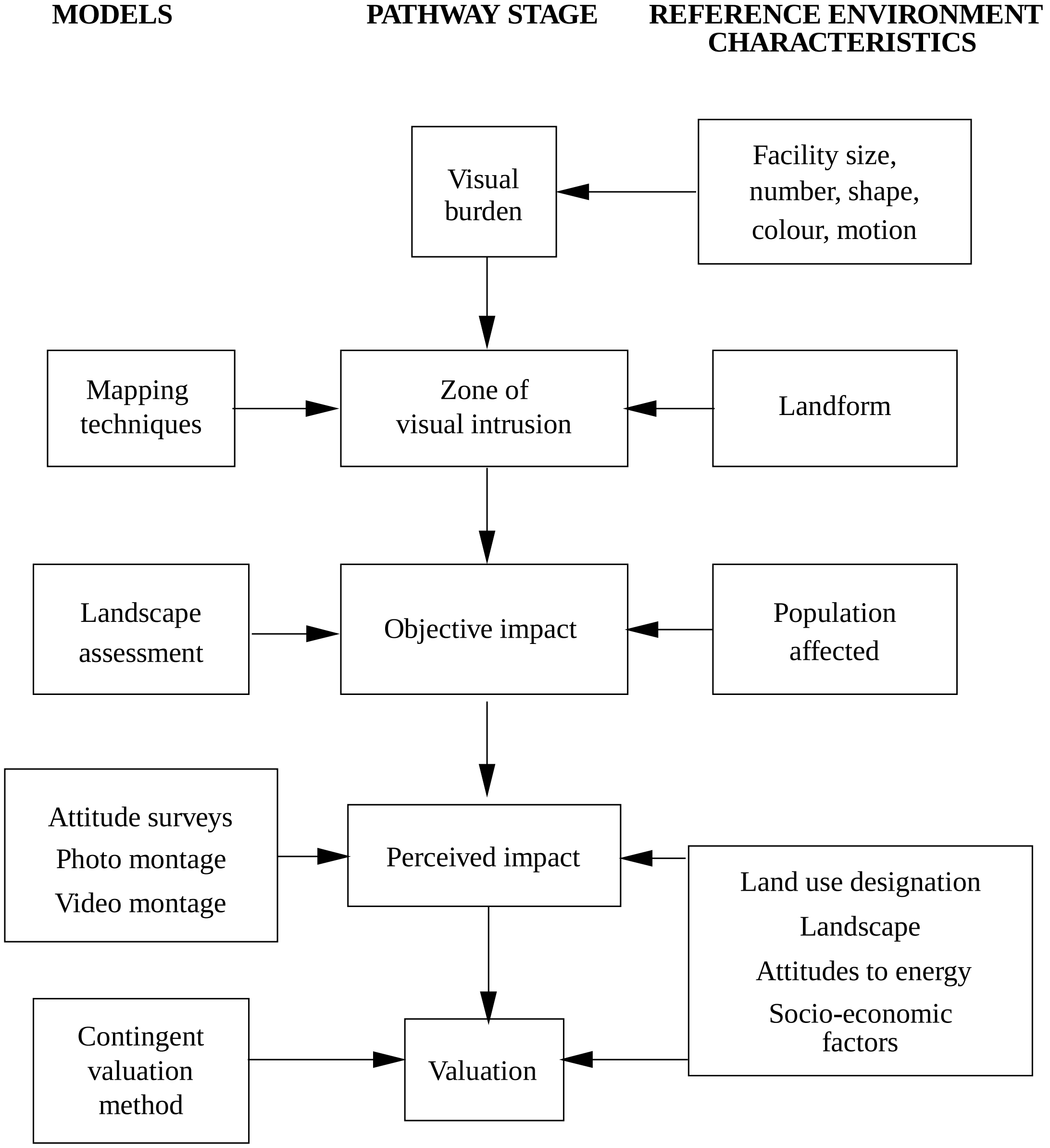

Figure 18.1 Impact pathway for visual intrusion.

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 landform 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 landform 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 18.1 attempts to show the range of factors that 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.

18.2.3 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 17, 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 and Garrod 1991). 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.

18.2.4 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.

18.2.5 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 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 landform. 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.

18.2.6 Landscape Assessment

It is inevitable that subjectivity is involved in landscape evaluation. However, there are concepts and procedures that 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 landform. This may be the most helpful approach in the development of strategic planning policies and estimation of the 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 18.1

Table 18.1 A classification of aesthetic factors in landscape assessment (Countryside Commission, 1993).

Category	Descriptors			
Balance	Harmonious	Balanced	Discordant	Chaotic
Scale	Intimate	Small	Medium	Large
Enclosure	Confined	Enclosed	Open	Exposed
Texture	Smooth	Textured	Rough	Very rough
Colour	Monochrome	Muted	Colourful	Garish
Diversity	Uniform	Simple	Diverse	Complex
Unity	Unified	Interrupted	Fragmented	Chaotic
Form	Straight	Angular	Curved	Sinuous

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.

18.2.7 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 vary 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.

18.2.8 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.

18.2.9 Aggregation

In ExternE it is assumed that the usual (and simplest) aggregation procedure can be used:

$$Aggregate\ damage = Mean\ WTP \times Affected\ \ population \qquad 18.1$$

The practical difficulty for energy sector development 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:

$$Damage = \left(N_R \times WTP_R\right) + \left(N_T \times WTP_T\right), \qquad 18.2$$

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 18.2 on a per unit area basis:

$$Damage / Unit\ \ Area = \left(N_R^{'} \times WTP_R\right) + \left(N_T^{'} \times WTP_T\right), \qquad 18.3$$

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.

18.2.10 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 a developer for tree planting or other screening efforts, there is therefore no internalisation. The damages calculated are therefore externalities.

18.3 Visibility

18.3.1 Issues

There is an apparent major disparity in attitude to effects of air pollution on visibility (or more precisely, visual range) between Europe and the USA. Well over one hundred pages of the NAPAP review (1990) were devoted to this subject, which has almost completely escaped attention in Europe. There is only one European study, Hylland and Strand (1983) which examines the effect of air pollution on visibility. In consequence almost all valuation data relating to effects on visibility are from the USA. Extrapolation to Europe is possible (as shown below) but prone to large uncertainty.

The main group of pollutants affecting visibility is the particulates, which can affect visual range, contrast and cause coloration. Visual range is commonly interpreted as the distance an observer would have to back away from an object in order for it to disappear. Energy production can effect regional visibility through gaseous emissions. Ideally one would like a valuation estimate in terms of visual range lost (in km), per unit of emissions but this is not possible to find.

The group on visibility at the Workshop on Benefit Transfer at Snowbird, Utah (June 1992, Association of Environmental and Resource Economists), after looking at the available literature, recommended using the percentage change in visibility and identifying a WTP in terms of such percentages. They also noted a problem that has been observed in several studies - i.e. that of isolating a WTP for visibility only and not including the WTP for the health effects of the same pollutants.

The approaches most used to value a change in visibility in a recreational or residential setting are the CVM and hedonic pricing. There are substantial difficulties that have to be overcome if successful visibility valuation is to be made. The environmental impact of reduced visibility is aesthetic in nature and is associated with a sensory experience and does not give physical effects on the body or on possessions. Thus, aesthetic impacts are to be differentiated from health, recreation or materials damage. The requirement of a sensory connection differentiates aesthetic values from non-use values based on knowledge rather than experience. However, it may be that people have psychologically intertwined the aesthetic value of air quality with the non-aesthetic benefits such as health, soiling and ecological effects. If this is the case, separating them may prove very difficult.

For a good CVM it is necessary to create a familiarity with the object in question. This may prove difficult to achieve for intangible aesthetic goods. Description of an aesthetic good is problematic since the very effort of capturing it in words or pictures may cause a respondent to value it differently than they would do based on their own prior experiences. The use of photographs can be of great importance here. It may be better to rank levels of visual air quality on a scale of say 1 to 10. There is also a problem of quantifying visual range. It is not easy to measure how far one can see. Individual perceptions may differ from scientific measurements of visibility, although according to some studies, there appears to be a linear relationship between the two.

Hedonic property pricing methods have also been used to measure visibility values. The method assumes that residents' preferences toward air quality will be reflected in the market price for homes and property. Hedonic analysis must rely on available data and since few aesthetic goods are readily measurable, the occasions when hedonic methods can be used are limited. Any proxies that might be used may be associated with other impacts; e.g. particulates are associated with respiratory problems and soiling.

The lack of interest in visibility as a pollution issue in Europe leaves us with only one European research study into its valuation, that of Hylland and Strand (1983). This study found it difficult to disentangle visibility impacts from health ones. They did not include morbidity effects in their survey, but a direct question asked at the end, revealed that a significant share of the respondents stated WTP was due to an expected health risk reduction. Only 14.5% stated that reduced visibility was the single largest negative effect of air pollution in the local area; 50.1% stated that health effects were the largest problem; 18.6% claimed bad conditions for raising children as most important (arguably belonging to the health risk category) and a final 16.8% reasoned that the smell and dirt/dust was the most prominent problem.

The study was conducted in the Grenland area of Southern Norway. People were questioned on a national (962 randomly sampled persons) and local level (1,004 randomly sampled persons) about their WTP to reduce the air pollution to the level in other Norwegian cities which will involve a 50% reduction in the annual number of days with the worst visibility level, as shown in a photograph. Nationally (as a single payment) a WTP was estimated at NOK 1060 - 1855 million (1990) (ECU 132.5 - 231.9 million) (0.6% of income tax). Local surveys revealed a WTP (as a single payment) of NOK 700 million (1990) or ECU 87.5 million. This translates to an annual payment of approximately NOK 900 (1990) or ECU 112.5 per person in the local affected area for a 'considerable' improvement in air quality. The national survey was crude with a very short and incomplete verbal description of the air quality improvement. The local survey used photos and verbal descriptions of the different air quality levels.

Commodity mis-specification and biases may be present. Use of income tax as a payment vehicle may lead to an understatement of WTP due to the already high income tax levels.

18.3.2 Methodology

The core reference used here is a review by Landrieu (1997). Light extinction results from two phenomena, scattering and absorption. The function proposed for the analysis adds together extinction from various fractions of ambient air;

$$b = b_{air} + e_S\left(SO_4^{--}\right)f\left(RH\right) + e_N\left(NO_3^{-}\right)f\left(RH\right) + e_O\left(organics\right)g\left(RH\right)$$
$$+e_C\left(elementalcarbon\right) + e_D\left(otherfineparticles\right) + e_G\left(NO_2\right)$$

b = light extinction coefficient of the atmosphere
b_{air} = scattering of light by molecules in unpolluted air = 0.011 km^{-1} (average value)
SO_4^{--}, NO_3^{-}, etc. are air concentrations of the pollutants of concern
$e_{subscript}$ defines the scattering efficiencies of each fraction. The units given in the Landrieu paper are $m^2\ mg^{-1}$, except for NO_2, for which e_G is expressed in $km^{-1}ppm^{-1}$). Values for each are as follows, noting in most cases that there is an absence of specific field data;

e_S	=	0.003	e_N	=	0.003
e_O	=	0.003	e_C	=	0.012
e_D	=	0.001	e_G	=	0.33

f(RH) and g(RH) are ratios of the scattering due to hygroscopic aerosols at a given relative humidity RH to the scattering at 0% RH

Ammonium is not specifically accounted for in the function proposed by Landrieu. We assume that all ammonium is present as either sulphate or nitrate.

Landrieu describes the complex interaction between humidity and scattering efficiency. Following Sisler *et al* (1994) the following is adopted for the average annual scattering effect of humidity on nitrates and sulphates;

$$f\,(RH) = 4.6 - 15(RH) + 19(RH)^2$$

and for the average annual scattering effect of humidity on organics;

$$g(RH) = 2.5 - 6(RH) + 5(RH)^2$$

Expanding the equation and expressing NO_2 in ug m^{-3} for consistency with the other pollutants, leaves the function;

$$\begin{aligned} b = \quad & 0.011 \\ & +0.003\left(SO_4^{--}\right)(4.6 - 15[RH] + 19[RH]^2) \\ & +0.003\left(NO_3^{-}\right)(4.6 - 15[RH] + 19[RH]^2) \\ & +0.003\left(NH_4^{+}\right)(4.6 - 15[RH] + 19[RH]^2) \\ & +0.003(organics)(2.5 - 6[RH] + 5[RH]^2) \\ & +0.012(elemental\ carbon) \\ & +0.001(other\ \ fine\ particles) \\ & +0.00017\left(NO_2\right) \end{aligned}$$

The ‘haziness index’ (dv) suggested by Pitchford (1994), can be combined with a value of 8 ECU/dv to quantify value (in ECU) per person;

$$value\ per\ person = 8 \cdot dv = 80 \cdot \ln \frac{b}{0.01}$$

Maddison (1997) has reviewed a number of different studies to produce an alternative estimate of WTP. One of the purposes of the analysis was to test whether the results of certain studies were statistically different from the rest of the literature. The function derived by Maddison, converted to give WTP in ECU, and omitting terms that covered studies that Maddison found to be flawed, was:

$$WTP = 125 \cdot \ln(V_2 / V_1)$$

V_1 = initial visual range
V_2 = final visual range

Overall the Maddison function gives lower damages. From the perspective of limited concern about visibility in Europe it would appear that this function is therefore preferable to that proposed by Pitchford (1994).

18.3.3 Recommendations

Since there is only one European study on visibility and it is unclear as to how much it is actually valuing visibility *per se*, it is recommended that this study not be used. If US studies are to be employed, uncertainties surrounding data transfer need to be fully recognised. The use of generic visibility estimates must be open to serious criticism, given that atmospheric conditions are different for different areas, as are population densities and personal preferences. A further issue is the periodicity of poor visibility episodes, given that most of the air quality data used in the study are generated on an annual mean basis that leaves us incapable of describing the frequency of episodes. In view of this, and for the lack of European data, it is recommended that no attempt be made to value visibility losses in the ExternE Study. It is, however, hoped that the presentation of a methodology in this report will stimulate further interest in this issue.

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19. NON-ENVIRONMENTAL EXTERNALITIES

Two separate areas were addressed under this heading in the Project; the externalities associated with nuclear proliferation and those associated with energy security. The following sections summarise the progress made, and are extracted from longer reports by Kuhn (1997) and Lockwood (1997).

19.1 Nuclear Proliferation

Proliferation of nuclear and dual-use technology has been on the political agenda since the 1950s and 1960s and is widely recognised as a major post-Cold War issue. The renewed concern for the dangers of proliferation has been triggered primarily by the recent experiences with Iraq's and North Korea's nuclear activities, the nuclear arms race between India and Pakistan, and rearmament planning in a number of countries in the Middle East and East Asia. Further, the disintegration of the Soviet Union reveals new threats from a nuclear superpower collapsing, whilst lacking a framework for effective export controls for nuclear and dual-use technology and for nuclear-related materials. Reports of illegal activities associated with nuclear materials, i.e. reports on dilemma posed by nuclear energy is that the same facilities can be used either for electricity generation or to manufacture fissile material for weapons. The difference between the two uses depends on how the facilities are configured and operated. Although all current reactor types could be the starting-point for seeking a nuclear weapon option, some reactor types, the fast breeder, the high temperature, and the graphite-moderated reactor are predetermined for nuclear proliferation. A determined nation can take either the plutonium, i.e. reprocessing route, or the HEU, i.e. enrichment route, or both routes in tandem to produce weapon-grade material in a clandestine programme. Until now the production of substantial quantities of HEU has required a large dedicated effort, whereas details of the plutonium path are today well known, and easier to hide. The operation of a relatively small nuclear reactor capable of producing plutonium, together with a reprocessing facility has therefore long been assessed as the most probable route for would-be proliferators. This remains a problem of utmost concern as developments in North Korea have shown. Yet, it is also necessary to be aware of all of the potential methods by which a nuclear weapon option may be pursued. This is the lesson learnt in the Iraqi case. Finally, taking account of the rapid technological developments, there has to be a shift in focus of intelligence resources, for example, from gaseous diffusion to laser isotope enrichment plants. About 10 countries are known to be substantially involved in laser enrichment research and development.

Proliferation of components of the enrichment and reprocessing technologies was the primary issue throughout the 1970s and 1980s. Export business and strategic concerns often played a more vital role than did international security concerns in the (non-) proliferation and export policies of several industrialised nations including Canada, France, Germany, the UK, the US, the former Soviet Union, and Switzerland. Under the guise of peaceful nuclear technology co-operation, these countries were willing to sell their nuclear technologies to nations like Argentina, Brazil, India, Pakistan, Israel, Egypt, and South Africa who had at that time refused to become members of the Non-Proliferation Treaty (NPT). Most of these countries are today suspected of having developed nuclear devices: indeed, India has just demonstrated so through a series of tests. Pakistan seems likely to follow.

Proliferation of know-how and of the fissile materials uranium and plutonium is an issue of special prominence in the past few years. There are no measures or regimes that can physically protect fissile material against misuse for weapons by nations or by groups. Inspectors of the International Atomic Energy Agency (IAEA), for example, have no technical means to detect unauthorised diversion of quantities of plutonium for weapons from commercial plants. The Nuclear Control Institute (NCI) in Washington, DC predicts that the year 2000 will mark the point in human history when more weapon-usable grade material begins circulating in civil commerce than exists in nuclear weapons. Several states, especially France, the UK, and Japan, have significant reprocessing programmes (involving their commercial use). The material is weapon-usable plutonium, created in civil reactors that generate electricity rather than in military reactors that produce fuel for weapons. The problem is that although the intended use of these two types of reactors is different, the by-product is the same – plutonium, an essential ingredient of nuclear weapons. The probability of theft and smuggling, and therefore, of misuse of fissile material for military or terrorist purposes as well as of accidents is increasing with the global inventory of separated plutonium. Thus, the added security risk from fissile material production facilities in the civilian nuclear fuel cycle, from using plutonium fuel in particular, is substantial. A prompt conversion of peaceful plutonium stockpiles into nuclear weapons is possible when there is a need for so doing.

The political situation in the mid-1990s

In the proliferation context, the analysis of technological developments is only meaningful and then most useful when discussed in the setting of international, regional, and national political developments and problems. The character of risk is changing with different global systems. In the post-Cold War era, for instance, the threats rather arise from regional hostilities and conflicts and the security vacuum in several parts of the world. States are increasingly more interested in and determined to create their own indigenous security infrastructures, even if undertaken with outside help. Then, there is also the fact that the technical barrier to the acquisition of nuclear weapons has steadily eroded with an increasing number of countries having access to the technology and thus possessing appropriate technological capabilities. For many countries the decision to acquire nuclear devices is, therefore, entirely political.

From a political point of view, there are several different nuclear hot spots in the post-Cold War world. The Middle East, South Asia, and North-East Asia can be regarded as the regions with highest proliferation concerns. Conflicts in these regions remain a major motivation for the states there to seek nuclear weapons, plutonium and HEU. Israel, for instance, and its unacknowledged nuclear weapons continue to be a provocation in the region. According to Albright et al.'s (1997) central estimates Israel possessed 460 kg, India 330 kg of plutonium, and Pakistan 210 kg of HEU at the end of 1995. The stocks of these *de facto nuclear-weapon States* are outside international controls and are believed to be part of their nuclear weapon programmes. India and Israel's stocks are even projected to grow. Although Pakistan is believed to have 'frozen' the production of HEU there are indications that it may start to produce unsafeguarded plutonium. Secrecy surrounding the Khushab reactor currently under construction suggests that Pakistan may have plans to separate plutonium there. *Iraq* is the only country in the world prohibited from possessing separated plutonium and HEU. Its pre-Gulf War facilities have been destroyed, yet, the country retains extensive expertise and ambition to reconstitute its nuclear weapons programme. *North Korea*'s nuclear programme is 'frozen' under the 1994 Agreed Framework, and progress is being made on building two light water reactors. The final outcome of this process will dep15 haws94(d9iTJ13.598s nuc)0w9(tion f)3.8(o)1.2(

It is the interplay of security concerns, matters of foreign and domestic policy, and commercial interests that make the effort to prevent the proliferation of nuclear technologies and materials an extremely complex affair. The nuclear technological capability is definitely a necessary condition for nuclear weapon proliferation to be possible, but not at all sufficient for proliferation to happen. Here many other military-strategic, institutional, political, economic, and social aspects play an essential role as well.

International non-proliferation regimes and policies

Several regimes regulate the non-proliferation of nuclear and dual-use technology. Most have been in existence for several decades, but are currently going through intense revision and evaluation due to the new security concerns. The Treaty on the Non-Proliferation of Nuclear Weapons (NPT) is the best-known element. The NPT came into force in 1970, was strengthened in the 1990s with the membership of Algeria (1995), France (1992) and China (1992), the latter two being declared nuclear-weapon States (NWS), and its indefinite renewal reached in May 1995. Several so-called 'threshold states' (states wishing to develop nuclear weapons and considered not far away from being successful), such as South Africa (1991), have also signed the NPT in recent years. Moreover, the years of negotiations on a Comprehensive Nuclear Test Ban Treaty (CTBT) were finally brought to an end in September 1996. So there have without doubt been some positive developments in the past few years which have reinforced the nuclear non-proliferation regime.

Overall national and international commitments to strengthen the traditional non-proliferation regimes and safeguard and control policies appear to have increased; they have especially been pushed in light of the disorders in the former Soviet Republics, and the Gulf War and its aftermath, but also by the rapid spread of the peaceful use of nuclear technology. For instance, several institutionalised groups who had not been active for some time were taking up their conventions and tightened several export regulations – the Nuclear Suppliers Group and the Australian Group in 1991, the Missile Technology Control Regime in 1993. Moreover, a great variety of new strategies and approaches about how to face and address the problem of weapon proliferation are formulated in both, theory and practice, by academics and policy makers alike. The Defence Counter-proliferation Initiative (DCPI) of the United States is in the first place a programme to develop new military capabilities to deal with the peril posed by the globalisation of nuclear high technology and commercialisation of fissile materials. Other concepts focus on a comprehensive and non-discriminatory curbing and cutting off of the spread of weapon-usable fissile material. They go as far as calling for a moratorium on programmes for the civil production and use of separated plutonium and HEU. On the other hand, the declared nuclear-weapon States and other supplier countries are still committed to their own nuclear and strategic programmes and plans, e.g. China and France to nuclear weapon tests, the United States to DCPI, Germany to Garching II, Japan to breeder programmes. This might adverse some effects and the progress achieved; it is at least casting doubt on the credibility of their non-proliferation policies.

Case studies on Iraq and North Korea

The two nations selected for detailed consideration during this phase of the ExternE Project – Iraq and North Korea – have received most attention in the context of international non-proliferation efforts in the post-Cold War world order. Both cases reveal similar characteristics. For instance, both countries are in parts of the world where the presence of (un)declared nuclear weapons make the regions – Middle East and East Asia – potential nuclear hot spots. But the two case studies also illustrate that there are many different ways for determined nations to proliferate and for the international regime to deal with their proliferation potential. Iraq was able to develop a significant nuclear weapon programme, for many years almost unnoticed by the international intelligence community, despite the fact that it had been a signatory to the Non-Proliferation Treaty (NPT), and despite regular on-site inspections by International Atomic Energy Agency (IAEA) officials. The amount of foreign assistance and technology that had fuelled Iraq's nuclear programme was truly staggering. In the case of Iraq, elements of counter-proliferation, that is active defence and pre-emptive attacks, were in some respect applied by the US and their allies. But only after years of special inspections carried out by the IAEA under United Nations (UN) mandate in the aftermath of the Gulf War was much of Iraq's nuclear proliferation potential uncovered and destroyed. The North Korean nuclear weapons programme has also been pursued in violation of Treaty commitments and in defiance of the international community. However, it seems to have been much more indigenous in nature also owing to the closed nature of North Korea. Only in the North Korean case, diplomacy and positive economic incentives were used for the solution of the crisis. Since hard evidence is still lacking in this case, the scope and stage of the North Korean weapons programme remains ambiguous and unclear to the international community even after its freeze – maybe not to the US intelligence community.

19.1.2 Possible Approaches for Integration into the ExternE Methodology

As uncertainties and unknowns are in the nature and complexity of the proliferation problem – it has not only a technological, but also a military-strategic, political, social, and economic dimension, external costs of nuclear weapons proliferation are definitely a sensitive and controversial issue. Both, damage estimates and probability assessment, can easily differ by orders of magnitude depending on the assumptions and judgements made and the approach chosen. So a profound qualitative examination and account of the global proliferation situation was crucial.

For the integration of the proliferation issue into the ExternE accounting framework, it would be necessary to quantify the marginal proliferation risk imposed on societies by a certain nuclear power plant in the European Union. In the following section, three approaches are presented which could deal with that high-uncertainty and low-probability severe-damages issue inside the ExternE accounting framework:

- a simple expected utility estimate in line with Shuman/ Cavanagh (1982)
- a more complex scenario approach considering the problem of regional hot spots

- an abatement cost approach
  (in this case damage and abatement costs can not always be strictly separated since abatement measures themselves may cause severe damages).

It should be kept in mind, that in the context of global risks, as for instance global warming, but also the nuclear proliferation issue, economists themselves have been questioning the applicability of the marginal cost approach. Pearce (1993) for instance writes: "An appraisal procedure that evolved from concerns with mainly localised and certainly marginal changes to the state of the economy is being called upon to apply to issues that are global in a non-marginal sense, i.e. significant changes in well-being are involved. A tool for fine-tuning decisions is being applied to contexts where fine-tuning is not the issue." Nuclear proliferation is not a market failure that can be corrected by individual consumers or producers acting in their private interests. The market failure almost certainly requires collective action to correct. Moreover, the criterion of site-specificity and marginality of external costs cannot be pursued for technical reasons – this should have become clear; the cost estimates would be average values only.

Proliferation damages and damage costs

Calculation of damage costs from nuclear proliferation was not possible in this study. Here, we review the issues, methods that have been proposed, and the very limited data already published.

Simple Expected Utility Approach

Shuman and Cavanagh (1982) sought to summarise the potential range of proliferation damages. For the upper limit they assumed that a global nuclear war would kill 10^9 people, with a cumulative probability of 10%. They then supposed that proliferation would increase this pre-existing probability by a tenth (1%) between 1982 and 2010. Charging the increment in expected deaths to the cumulative use of nuclear power worldwide between 1982 and 2010, these assumptions translate to 165 to 239 expected deaths per GigaWatt-year. Adopting a value of statistical life of US$3,000,000, Shuman and Cavanagh concluded with an upper limit of 112.6 mills_{1982}/kWh. They also provided a lower bound cost for proliferation of zero because of the speculative nature of the analysis. Shuman and Cavanagh themselves were fully aware that their range of 0 to 112.6 mills_{1982}/kWh was highly controversial because of enormous uncertainties and unknowns.

If such an approach were to be adopted (noting that it was not in ExternE, because the study team felt unable to address the uncertainties involved in a meaningful way), it would be necessary to take into account the changed political framework of today, that is the post-Cold War world order.

In view of today's world order, this method would be pretty arbitrary. At least the probability assessment needs to be based on expert judgements or some kind of meta-analysis. If the risk-based damage cost approach of Shuman and Cavanagh was pursued in a more detailed analysis, the project would indeed be extensive, but also more appropriate for the complexity of the issue and today's political structures. It must be borne in mind that often a decision by a state to seek the development of nuclear weapons, or for that matter its success in the endeavour, will depend predominantly on non-technical considerations. Such factors will vary considerably from one country to another and will depend upon the capabilities of organisation and management associated with the complex, long-term commitment of scientific and technological expertise and infrastructure, the ability to keep the operation covert, the type and extent of foreign trade and business contacts, hard currency supply, as well as the costs, both domestic and international, of the effort being uncovered at some stage. This supports the idea of an analysis on a country-by-country basis.

Risk assessment considering the problem of nuclear hot spots

Following the idea of a country-specific analysis, the first necessary and important step for the projection of the possible spread of nuclear weapons in a region or country is the assessment of the *nuclear capabilities* of each country through the analysis of its present nuclear programmes. The civil nuclear fuel cycle – type of reactors, enrichment and reprocessing plants, stock of fissile materials – and military applications of the nuclear capability – fissile materials, weapons fabrication, delivery systems – need to be analysed. Based on this information and a projection of the plutonium production capacity the weapons production potential and the statistical probability of future use can be estimated. Secondly, a country's *incentives and disincentives* for present and future nuclearisation have to be assessed in the national, regional, and international context. This could be done with the help of a country-specific qualitative cost-benefit analysis. One can for instance assume that near-nuclear countries are likely to decide to go nuclear soon if their probable total benefits outweigh the expected total costs of their nuclear weapons development programmes. For each region or country a series of *scenarios* of possible outcomes must then be developed depending on the influencing variables chosen and assumptions made. The scenario method is often used in political science and is also common in the proliferation literature. Yet, *probabilities* are usually not attached to the respective scenarios, the impacts and damages are not assessed. Hayes (1991) for example evaluated the potential for a peaceful resolution of the nuclear dilemma in Korea. He states that "whatever their probabilities, these outcomes are conceivable." The outcomes should not be regarded as exclusive, either. One scenario could evolve out of, or overlap with, another. For North Korea, Hayes came to the general conclusion that the price of political stability and a non-violent resolution of the nuclear dilemmas is small compared with the potential costs of nuclear proliferation and the related arms racing and enhanced risk of war, with all the attendant social and economic costs of the continued division of Korea.

This country-specific approach seems to best comply with the ExternE accounting framework. Even if it were only applied to the nuclear hot spot countries listed before, the necessary research effort would be high. The fact that probability and damage assessments are not widespread in the relevant literature makes the type of approach proposed here much more difficult. It cannot be based on a 'simple' literature review, but would require new, interdisciplinary research. As a result, it might be possible to quantify some aspects, whilst others will turn out not to be quantifiable.

In general, pursuing a risk-based approach – the multiplication of probabilities and impacts to one damage value – for nuclear energy raises several problems. Similar to the case of severe nuclear reactor accidents, the discrepancy between the very small probability of proliferation damages to occur and their large scale is extreme. Therefore, a single risk value for the impacts of proliferation would among other things conceal that many possible damages are irreversible and that the costs are orders of magnitude higher than for other impacts with the same cost value. It would be misleading to directly compare the risk-based estimates to estimates of other damage categories and other fuel cycles, since the applied unit of measurement is different. In such a case where one or both factors have an extreme value, the German Advisory Council for Environmental Issues suggests that evaluation and decision is based on an independent assessment of probabilities and damage costs.

Abatement and Control Cost Approach

One could argue that the damages of a worst-case scenario ('maximum regret'), that is the damages of a global or regional nuclear war – the former appearing almost impossible in today’s world order – would be irreversible and/ or so far-reaching that one should put every effort in preventing this catastrophic event with minimal costs. An abatement cost approach is more appropriate in this case. It might be pursued to get a rough comparison of the impact relative to other impacts of the nuclear fuel cycle and to partly take account of the external costs of nuclear energy due to proliferation. The sum of control and abatement costs would most probably be orders of magnitude below the damage costs of proliferation, except in the case of the deployment of military counter-proliferation measures where wars can also be waged and people get killed or contaminated – as in the Iraqi case.

Examples of areas in which money is spent today to control and prevent proliferation are IAEA safeguards costs. Valuable information of relevance to proliferation is also being collected and analysed by organisations ranging from national intelligence agencies, some with billion dollar budgets, to the media and other interested groups. The list can be expanded as long as you like; the total would only be a fraction of the damage costs of a nuclear war.

The costs of the Gulf War and the special IAEA inspections in Iraq in its aftermath and the expenditures of the United States and other countries to freeze and replace North Korea's nuclear programme would fall within the abatement cost category. In the Iraqi case, the distinction between damage and abatement costs might not be apparent at once since the damages of the Gulf War have been severe and the prevention of proliferation was not at minimal costs. However, the most effective appeal to convince the American Congress and public that the US should use military force against Iraq proved to be the contention that Iraq might soon acquire nuclear weapons. So it has in fact been the prevention of nuclear proliferation that has been a main, but not the only reason for the military conflict between Iraq, the US, and their respective allies causing injured and dead people (estimates vary between 25,000 and 200,000 human losses), and many other damages. Attribution of these casualties to the nuclear fuel cycle is not possible because the causes and consequences of the Gulf War were so complex. and varied.

As a result, the quantification of impacts of the Gulf War does not seem to be a very serious or desirable scientific exercise in the external cost of proliferation context. The Iraqi case may, however, at least show that the external effects of proliferation are significant when compared with severe reactor accidents.

Multi Criteria Analysis

Even though costs and economic efficiency are important criteria of political decision making, it must be acknowledged that the proliferation problem comprises more aspects. A multi-dimensional assessment scheme is therefore required. However, a final set of criteria which is unbiased, consistent and non-controversial only exists in an ideal, static world. In the real world, criteria and targets for political decision-making is a matter of discussion and change over time. They need to address aspects such as reversibility/flexibility, feasibility, environmental and social issues; these aspects may be as important as economic aspects.

Within such an extended assessment scheme, physical impact assessment (e.g. environmental and health impacts) and cost estimates may both remain important elements.

19.1.3. Conclusions

1. At the present time it is not possible to make robust estimates of damages linked to nuclear proliferation.
2. There is limited literature in this area dealing with quantification. However, this section has briefly reviewed possible methods for analysis.
3. It is hard to see how the broad range given by Shuman and Cavanagh (1982) could be usefully applied in decision making. However, by seeking to provide a quantified estimate the Shuman and Cavanagh paper did at least start to advance the debate, and raised a number of useful issues.
4. Further work in this area is clearly desirable to promote transparency in response to the problem of proliferation.

19.2 Energy Security

There were two main objectives in this part of the study; first, to critically evaluate earlier studies of energy security externalities undertaken in the previous phase of the ExternE Project, and second, to extend and improve on these externalities wherever possible.

An energy security externality in the electricity supply industry (ESI) arises when a decision made by electricity suppliers (capacity choice, fuel mix, etc.) has an effect on the price or availability of electricity (or on the price or availability of fuels that are used to generate electricity), which is not internalised, or taken account of, by the electricity supplier. An important and topical example relates to non-fossil fuels. It is often argued that increased electricity generation from non-fossil fuel will reduce dependence on imported fossil fuel, and therefore reduce the overall variance of fuel prices, and/or reduce the probability of fuel rationing. Energy security externalities are subtle and difficult to identify, and for this reason, earlier analysis in the ExternE Project in this area suffered from two main limitations. First, they tended to focus on particular externalities, and second, the methodology used was not always well integrated with the 'standard' impact pathway methodology developed for the study more generally.

The work undertaken in this task attempted to address both these major problems. First, a comprehensive framework for classifying energy security externalities was developed. Three kinds of externalities were identified: monopsony wedge, incomplete rent capture and macroeconomic externalities. Each of the main decisions made by the ESI (choice of new capacity, changes in the fuel mix (merit order), fuel stockpiling and hedging on fuel futures markets) may give rise to one or more of these externalities.

The 'impact pathway' for each of these three externalities was then studied. The impact pathway is not a physical one, as it is for environmental externalities, but operates through markets. The main implication of this is that 'impact pathways' are identified not by reference to the scientific literature (as they are for say SO_2 pollution), but by constructing models of the national economy, and tracing the effects of (say) an increase in demand for oil by the ESI on other economic agents by formal analysis of the model.

This formal analysis can be done in two ways. If the model is sufficiently simple, the impact pathway(s) can be traced analytically (i.e. through algebraic manipulation of the model). If the model is complex, the impact pathway(s) must be traced by computer simulation. In either, case, once the impact pathways have been studied, the next step is economic valuation. This is usually quite straightforward, as the end of the impact pathway is a change in price or in the change in consumption of a marketed commodity. On the other hand, to get ‘realistic’ estimates of the external costs on a mECU/kWh basis, it is necessary to calibrate the model, i.e. to choose ‘realistic’ values for all model parameters. This can be difficult and time-consuming.

This was beyond the scope of the current project. Hence, the main emphasis was on studying the impact pathways of the various externalities using a collection of very simple models of the national economy, which was simple enough to be studied analytically. This approach gives us some qualitative information; in particular it enables us to assess whether an externality is positive or negative (or whether it cannot be signed).

One partial exception is in the case of the monopsony wedge externality. Here, both Lockwood (1995) and Canon and Smeers (1996) obtained estimates of the monopsony wedge, ranging from 0.075 mECU/kWh to 9.76 mECU/kWh, though both analyses were subject to significant limitations.

Macroeconomic externalities are shown to be unambiguously positive, whereas this cannot be said for incomplete rent capture externalities. Moreover, the latter may be (partially) internalised by explicit or implicit payments to cover the cost of capacity, as are used in the UK.

An alternative approach to quantifying energy security externalities (ESE) was studied following the work of Sanchez (1995). Sanchez did not attempt to identify externalities arising from uncertainty about price or supply of imported fuel, but tried to estimate the gross cost to the national economy of this price or supply unreliability. This gross cost then provides an upper bound on the ESE. Although this method does not identify the externality, it has a major advantage that gross cost is much more easily estimated, as the impact pathway does not need to be identified; all that is required is to estimate the fraction of the national income that is required to compensate households for a unit reduction in exposure to fuel price risk. A number of different estimates of this fraction, based on a calibrated model, are presented in the full report. These can be used to calculate the benefits, on a mECU/kWh basis, of an increase in the diversity of the electricity generation portfolio. A notable finding is that this fraction may be negative, i.e. households may prefer to be exposed to fuel price risks. Moreover, sensitivity analysis suggests that this fraction is highly sensitive to model parameters whose values are not well known, e.g. risk aversion parameters. In spite of this sensitivity, however, the simulations indicate that the upper bound on the ESE is small in absolute magnitude, i.e. not more than 1.5 mECU/kWh.

19.3 References

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20. AGGREGATION OF EXTERNALITIES DATA

20.1 Introduction

During discussions with potential users of external costs it was apparent that there was a need for aggregated information to be used for na

20.2 Simple Aggregation Methodology

20.2.1 Fossil Fuels

Global Damages

To convert damage values in ECU/t (see Chapter 6 for the values recommended in the study) to the total electricity sector damage, it is necessary to multiply by the emissions from the power sector in tonnes. Emissions should be taken from national statistics for the whole power sector for the relevant member state (upstream fuel cycles emissions should ideally be added, but these are a second order correction at least for carbon dioxide). If these data are not available, emissions of carbon dioxide from each fossil fuel may be calculated approximately by multiplying fossil fuel inputs in TJ by the following factors: coal 87 t/TJ, oil 73 t/TJ, gas 50 t/TJ.

Regional Scale Damages

Regional scale damages should be derived by aggregating from analysis of individual fossil fuel power plant(s) in the country of interest. In this instance regional scale damages are those of SO_2, NO_x and PM_{10}. It should be recalled, and noted in any assessment, that reliably quantified damage in ExternE exclude impacts to forests, fisheries, natural ecosystems and buildings of cultural value. For treatment of ozone damages see Chapter 16.

To assess the total damages due to national emissions from the electricity sector, the specific damage values (ECU/t) for each pollutant should be multiplied by the national electricity sector emissions. This assumes that the reference power plant values are transferable within the country. For low stacks there may actually be significant site sensitivity, especially for particulates emitted in an urban environment. However, for high stacks the sensitivity is small, as the range of these pollutants exceeds the size of all European countries. The approximation is therefore reasonable for the electricity sector where high stacks predominate.

National emissions should be taken from national statistics for these emissions. It is important, in this case, not to use the g/kWh emission factors from the reference power plants typically considered in the ExternE project as a proxy for the whole sector as the new plant with which the study has mainly been concerned, is very unlikely to be typical of the sector as whole.

Local Impacts

Local impacts are more difficult to aggregate sensibly as they are inevitably more site dependent. However, they are, for the most part less important, so accuracy is less critical to the overall sectoral damage assessment. In most cases therefore all that is required is a linear extrapolation of existing results. For transparency of course the analysis must show which damages have and have not been quantified.

20.2.2 Nuclear

Reliable values for accidents, high level wastes impacts, nuclear proliferation and impacts of terrorism have not been developed in ExternE. These omissions may well be significant and therefore should be clearly noted in any assessment.

ExternE results show that nuclear damages are characteristic more of the technology than the site, and therefore aggregate damages should be assessed by scaling up damages most typical of the technologies used in each stage of the fuel cycle. Usually, but not always, the national implementation in the Member State will be the best baseline data source. Where national implementations are not available, the ExternE published results for France should be used, except adjusted as explained below to take account of technological differences.

The recommended adjustments are as follows:
- where reprocessing is not used, the ExternE damages due to the reprocessing stage should be excluded,
- where reprocessing is undertaken other than in the French UP3 plant assessed by ExternE, global average carbon-14 emissions from UNSCEAR should be used instead. The effect is to multiply the long-term global damages by a factor of 3.2. An exception is the UK THORP reprocessing plant, which has carbon abatement, giving a factor of 0.1.
- for reactors other than PWRs, the generation stage damages should be modified to reflect the higher emissions of carbon-14 documented by UNSCEAR. The result is to multiply

Mining and milling

The mining/milling collective dose calculated by ExternE for the French implementation is split roughly evenly between the regional and local doses. The local collective dose is 0.085 man.Sv/TWh, although this is strongly dependent on the technology assumptions made and UNSCEAR assumptions are probably better for most other sites. For the Lodève site, the population density in the local area (within 100 km) is 1.96 million, equivalent to a population density of 62 per km^2 (close to the European average). By world standards, this is quite a high population density, implying that local collective doses for most other areas' sites will be significantly smaller.

If aggregation calculations are based on the ExternE report values, a reasonable approximation of the local collective dose, compared to the Lodève site, will be achieved by scaling the ExternE number with the local population density. If, on the other hand, mining/milling doses were to be calculated from UNSCEAR data to represent more typical mill tailings management techniques, the scaling would need to be based on the UNSCEAR assumption of 3 persons per km^2 in the local area. At this population density, the local collective dose is likely to be small compared to the regional scale impact, although UNSCEAR-93 does not give separate figures for the two ranges.

For most world uranium sources, the UNSCEAR assumptions about local population density (as well as mill tailings management) are likely to be better than simply transferring those of the French fuel cycle. At low local population densities, the local impact is small compared to the regional scale impact, and therefore small changes in the local population density will not greatly affect the total collective dose. The UNSCEAR values can therefore be used unamended.

High Level Waste Disposal

In the ExternE assessment, the principal dose from the HLW disposal site within 100,000 years is due to technetium-99 in drinking water. Assuming that the PAGIS analysis on which this result is based would be valid for any other geologically appropriate site, only this pathway needs to be considered in transferring the results. The local impact in the ExternE assessment of 0.025 mECU/kWh (at 0% discount rate) was based on an assumed constant population of 7000 using drinking water at the Auriat site. Transferability is therefore rather straightforward. If the local population is known, the ExternE result should be scaled according to the ratio of the local population to 7000.

In general, however, the final disposal site for HLW will not be known, and therefore the potentially affected population is unknown. In the absence of better approaches, the results from the Auriat site may be used unmodified.

Conclusions

It is concluded that whilst, in principle, adjustments need to be made to local scale impacts to account for population density, in practice the results of the ExternE assessment in France (replaced by more general UNSCEAR assessments for mining and milling for other uranium sources) may be used generally in aggregation.

20.2.3 Renewable Energy Sources

In general, renewable energy sources have rather low damages, and those tend to be rather localised and short term. The result is that renewable energy external costs are rather site dependent, so that aggregation is difficult, but probably the aggregate damages are not significant in the overall energy system in most countries.

20.2.3.1 Wind

A model for transferring the costs of wind noise is discussed below. Ideally this is the route to calculating aggregate wind power noise. However, it requires information on population density, source and background noise level. If these are not easily available, it is suggested that a noise externality of 0.1 mECU/kWh be assumed, which is typical of wind farms analysed to date. This should be multiplied by the total wind energy generated (in MWh/year) to give the aggregate damages in ECU/year.

There is no reliable method for quantifying the external costs of visual impacts of wind energy - certainly not one that is transferable across the range of projects, landscapes and cultures involved in the EU. Visual damages cannot therefore be included and this should be noted in the assessment.

Quantification of noise externalities from wind farms

Noise external costs have been estimated for a few technologies in a few locations within the ExternE project. The analysis here uses a simplification of the methodology derived for noise from wind farms (see Chapter 17) to investigate the key parameters affecting noise externalities from a single source. The resulting simple relationships can be used to transfer noise damages from one site and source to another location. It is suitable for point noise sources, like power plants, but not for extended noise sources, such as transport routes.

These calculations use a simple model of noise dispersion in two dimensions, pay no regard to barrier attenuation (i.e. the shielding effects of natural or artificial barriers) or to meteorological conditions, such as inversions. The results are therefore only strictly valid for open land, average weather conditions, flat terrain and a uniform population density in the affected area. For more complex instances or more accurate calculations it is necessary to assess noise impacts and damages from first principles (European Commission, 1995). The approximations will tend to over-estimate noise, giving an upper limit to the noise damages.

The increase in noise, and therefore the amenity damage, is calculated at all receptor locations and integrated over the affected area. Because of the rapid reduction in noise as a function of distance from the source, the area affected significantly typically extends only about 1 km from the source. A uniform population density is assumed in this area. The integral converges rapidly and produces a result for the noise amenity damage with the following form:

$$f(RH) = 4.6 - 15(RH) + 19(RH)^2 \qquad 20.1$$

where

D is the damage in kECU/year,
S is the source sound power in dB(A),
B is the background noise in dB(A),
H is the household density in households per km^2,
V is the amenity value of noise in kECU/year/household/dB(A), and
f is a function of the difference between the source sound power and the background noise.

The damages may be converted to the functional unit of mECU/kWh by dividing by the annual electricity output of the power plant. For a power plant with an annual output of 1 GWh/year, the result is numerically equivalent to the damages in kECU/year. In other cases, the damages in mECU/kWh are obtained by dividing the numerical result above by the power station output in GWh/year.

The effects of population density and value of amenity in Equation 20.1 are rather straightforward. The noise damage is proportional to both. The effects of source and background noise are more complex. Some calculations at 5 dB(A) intervals are presented in Table 20.1. These allow transfer between different noise levels.

If the sound source power is known, but the noise from it measured at a given distance is known, the sound power, S, can be reconstructed from the measured noise, M, at distance, x, as follows:

$$S = M + 8 + 20\log_{10} x + 0.05x \qquad 20.2$$

The effect of household density depends on the noise increment at that point. Ideally, the calculation should integrate the product of a variable household density and the noise increment over the affected area. In practice, Equation 20.1 has been derived from simple calculations with constant population densities. For more complex calculations it is necessary to undertake individual calculations using the methodology described in Chapter 17 and elsewhere (European Commission, 1995). The results with constant population density may be assessed either over the whole affected area, or excluding a circular area around the source, where there is no habitation. The latter case is more likely for power plant and most other industrial facilities. In either case, the damages are proportional to the population density. Both are shown in Table 20.1.

The damage cost is likely to be insignificant, even for a small power plant unless the source noise exceeds background noise by at last 30 dB(A), even in a fairly densely populated area. For higher noise levels, damages may be significant. Scaling factors for transferring results from one source or background to another may be derived from the ratios of damages for different noise levels.

The results in Table 20.1 apply to a single source. In practice many power plants will have different sources which need to be considered additively. Where the sources are a distance apart that is large compared to the distance from affected observers, the use of Equation 20.1 is no longer valid and more complex calculations are required. This is likely to be the case for most large wind farms, where there will be relevant noise receptors (houses) within distances of turbines that are comparable with the distances between turbines. Where sources are relatively close together, their effects are additive. Because of the nature of the dB(A) scale, addition should be undertaken using Equation 20.3.

$$N_{total} = 10.\log_{10} \sum 10^{N_i/10} \qquad 20.3$$

where N_{total} is the total noise due to individual sources N_i.

20.2.3.2 Hydropower

There is a wide range of types of hydropower scheme, notably in terms of size and location. The external costs can therefore be expected to vary significantly. For modern small run-of-river schemes outside recreationally important areas, it seems likely that the damages are small. A larger modern scheme in Norway was found in ExternE to have aesthetic damages of 2 mECU/kWh. In the absence of better data it is suggested that provisional values of 2 mECU/kWh for large schemes and zero for small schemes be used in aggregation. Aggregate damages should be calculated by multiplying these by the total hydropower generation. The provisional nature of the assessment should always be noted.

20.2.3.3 Others

The methods described above will allow a simple aggregation to be followed for the major fuel cycles used in the European Union. For some of the fuel cycles not discussed explicitly (e.g. biomass) estimates can still be made of their likely most important effects. ExternE data does not yet allow reasonable estimates of aggregate damages from some other renewable electricity sources, as there is a multiplicity of different technologies and fuels. However, in most EU countries these technologies have a small share of aggregate electricity generating capacity, and therefore this omission is not serious. However, it should of course be noted in any aggregate assessment.

20.3 Multi-source EcoSense Aggregation

20.3.1 The Multi-Source Model

The more detailed approach was implemented for the German and UK power sectors using an extended *multi-source* version of the EcoSense software. As a first step prior to air quality modelling and impact assessment, the multi-source version of EcoSense supports the definition of any European wide emission scenario by using a link to the CORINAIR database. In a subsequent analysis, differences in concentration levels and environmental impacts between different scenarios are evaluated within the system.

The CORINAIR database provides emission data for a wide range of pollutants according to both a sectoral (SNAP categories) and geographic (NUTS categories) dissaggregation scheme (Table 20.2). A transformation module implemented in EcoSense supports the transformation of emission data between the NUTS administrative units (country, state, and municipality) and the grid system required for air quality modelling (EUROGRID or EMEP). Based on this functionality, a user can change emissions from a selected industry sector within a specific administrative unit, create a new gridded European-wide emission scenario taking into account the previously specified modifications, and compare environmental impacts and resulting external costs between different scenarios. Following this approach, the CORINAIR 1990 emission data was used as the reference scenario, and additional scenarios were created by setting emissions from the German and UK 'public power and cogeneration plants' (SNAP category 0101) respectively to zero.

Table 20.2 CORINAIR sectoral and geographical dissaggregation scheme (examples)

SNAP Categories		NUTS Categories	
01	**Public power, cogeneration**	**R**	**EUR 15**
01 01 01	Combustion plants >= 300 MW	**R1**	**Germany**
01 01 04	Gas turbines	R18	Baden-Württemberg
03	**Industrial combustion**	R181	Stuttgart
04	**Production processes**	**R2**	**France**
04 01 01	Petroleum products processing	R21	Ile de France
04 03 01	Aluminium production	**R5**	**Belgium**
05	**Extraction/Distribution of fossil fuels**	R5300	Brussels
07	**Road transport**		
11	**Nature**		
11 08	Volcanoes		

The implementation of the approach described depends on the availability of CORINAIR emission data, which is currently available for 1990 (data for 1994 are expected to be published shortly). Due to major changes in the electricity industry since 1990, in particular in the German ‘neue Länder’, results derived from 1990 data do not entirely reflect current conditions. Expected changes in the results due to the use of more up to date data are discussed. CORINAIR 1990 does not provide data on particulate emissions. For the present analysis, particulate emissions from the power sector were taken from other sources. As particulates are considered as non-reactive pollutants, there are no problems arising from this procedure.

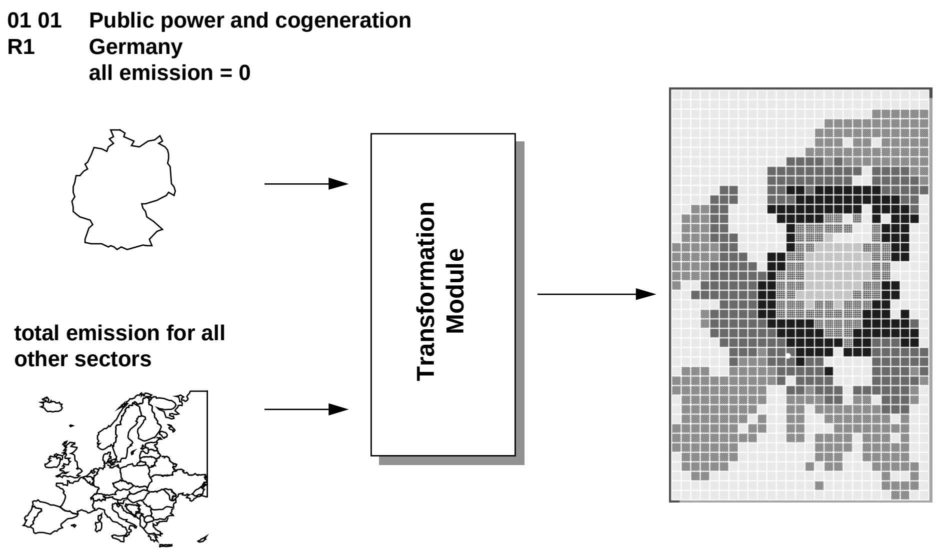

Figure 20.1 Generation of Emission Scenarios

20.3.2 Aggregation Results for Germany and UK

An example of the change in concentration levels and the spatial distribution of impacts calculated using the multi-source EcoSense is shown in Figure 20.2. This clearly shows the high concentrations of SO_2 resulting from the old coal burning power plants in ‘neue Länder’. Damage costs resulting from the operation of fossil fired public power and cogeneration plants in Germany and the UK in 1990 are presented in Table 20.3, while Table 20.4 shows aggregated damage costs per tonne of pollutant emitted.

Table 20.3 Damage costs in million ECU from fossil fired public power and cogeneration plants in Germany and the UK in 1990 (Emission data from CORINAIR 1990)

	Germany	UK
SO_2, NO_x, PM_{10}		
Health effects		
mortality [1]	28027	19631
morbidity	2987	1736
Crops	247	146
Materials	509	895
Ozone impacts (aggregated)	617	1164
Global warming		
low	1512	756
mid 1%	7164	3564
mid 3%	18308	9180
high	55322	27540

[1] Valuation of mortality is based on the YOLL-approach, 0% discounting.

Table 20.4. Damage costs per tonne of pollutant emitted (excluding ozone damage)

	SO_2		NO_x		PM_{10}	
	Change in annual emissions (in kt)	Damage costs per tonne SO_2 (ECU/t)	Change in annual emissions (in kt)	Damage costs per tonne NO_x (ECU/t)	Change in annual emissions (in kt)	Damage costs per tonne PM_{10} (ECU/t)
Germany	- 2232	9732	- 411	4214	- 477	18655
UK	- 2729	7397	- 776	2332	- 27	16934
West Burton	+163.0	8135	+29.0	5413		18730
Lauffen plant	+ 2.2	13676	+ 2.2	15684	0.55	23857

For the purpose of comparison, Table 20.4 includes results that have been calculated previously for the Lauffen and West Burton coal fired power stations. While the damage costs per tonne of PM_{10} are similar in all four cases, the damage costs per tonne of SO_2, and in particular per tonne of NO_x show surprisingly high variations, especially with respect to the German reference site. These differences are explained by the spatial variations in SO_2, NO_x and NH_3 emissions, which strongly influence the formation of sulphate and nitrate aerosols. The difference between the aggregated damage values for the UK and Germany and those for the individual power stations indicates the level of inaccuracy that would be experienced by extrapolating damage costs from individual power stations to the entire power sector.

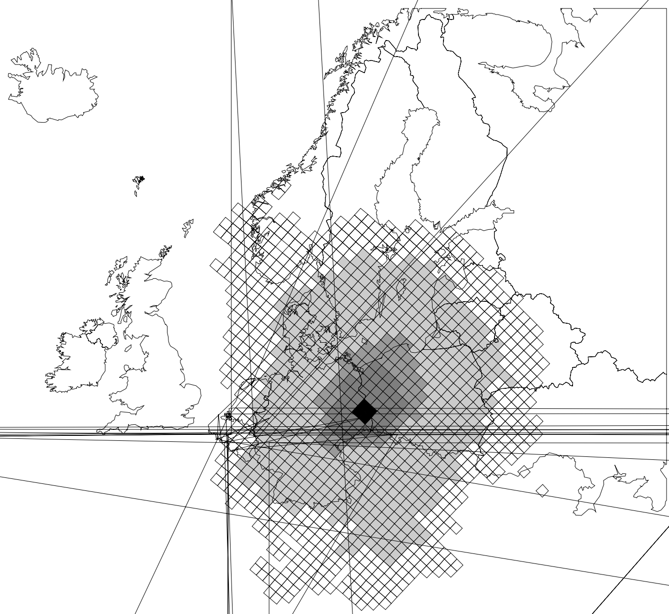

Table 20.5. Public health, agriculture and material damage costs associated with electricity generation from fossil fuels in the UK in 1995.

	SO_2		NO_x		PM_{10}	
	Emissions (kt)	ECU	Emissions (kt)	ECU	Emissions (kt)	ECU
Coal	1 481.26	1.10E+10	431.75	1.01E+09	33.00	5.60E+08
Oil (peak)	71.60	5.30E+08	30.00	7.01E+07	0.87	1.47E+07
Natural Gas	-	-	23.33	5.43E+07	-	-
Orimulsion	73.20	5.41E+08	8.88	2.07E+07	0.13	2.21E+06
Total	1626.06	1.21E+10	493.96	1.16E+09	34.00	5.77E+08

The damage costs from the fossil fuels are dominated by mortality and global warming impacts, which both are impact categories related to considerable uncertainties. The influence of different valuation concepts (*Value of Life Year Lost* versus *Value of Statistical Life*) and different discount rates were addressed to show the potential range of results.

Aggregated damages for the nuclear power plant were estimated based on the results obtained for the assessments of the nuclear fuel cycle in Germany and the UK. Results for renewables in the UK were estimated based on the previously published ExternE results for the wind and hydro fuel cycles. The externalities associated with renewables are very site specific and damage estimates have been made based on some major assumptions concerning the transferability of data. In Germany the major source of renewable energy is small scale run-of-river hydro. This has not been assessed in the German National Implementation project however results from other countries suggest that associated externalities are close to zero.

Results in terms of total damage costs per year and average costs per unit electricity for the German and UK power sector are summarised in Table 20.6.

Table 20.6. Damage costs from the German and UK power sector

	Mill. ECU/a		mECU/kWh	
	min	max	min	max
Germany (1990)				
Fossil fuels	41337	95140	23.6[1)] 373.3[2)]	151.3[1)] 579.6[2)]
Nuclear	139	1037	0.91	6.8
Others	n. q.	n. q.	n. q.	n. q.
UK (1995)				
Fossil fuels	15124	36932	64	156
Nuclear	196	338	2.2	3.8
Others	12.5	12.8	1.7	2.3

1) former Federal Territory
2) ‘neue Länder’

To report mean values for Germany would be misleading due to the very high emissions in the former GDR (‘neue Länder’). Hence, separate results are presented for fossil fuels for the former Federal Territory and the ‘neue Länder’. While emissions from the power sector in the Former Federal Territory have been constant or slightly increasing over the last years, there has been a drastic reduction of emissions in the ‘neue Länder’, leading to a reduction of the overall damage costs of about 25 % since 1990. The quantified damage costs from nuclear power generation are relatively low compared to the fossil fuels. Although nuclear energy contributes to nearly 30% of power production in 1990, the share of damage costs from nuclear electricity is only about 1.5%, according to the assumptions made in this study. Note that alternative assumptions on methods of radiological waste disposal, and on valuation of major accidents could affect this result significantly.

The results for the UK are based on 1995 emissions. The fossil fuels have been calculated using the damage/tonne of pollutant calculated using the multi-source EcoSense, scaled using 1995 emissions from the UK power sector. As discussed previously, these results are substantially lower than those for 1990 due to changes in the technology mix. The results for nuclear have been estimated based on the results obtained for the UK nuclear fuel cycle and the renewables have been estimated from the results for the wind and hydro derived in the previous phase of the ExternE project.

20.4 Conclusions

This Chapter presented the first aggregation study based on the detailed bottom-up impact pathway approach. From the results obtained the following conclusions can be drawn:

- Total damage costs from the power sector in Germany and the UK are 1-2 % of GDP.
- There is a significant variation in damage costs per tonne of pollutant emitted across countries, and - perhaps more important - even within a country. The simple aggregation method based on transfer of pollutant damages calculated for individual plant can thus lead to significant error. The error can be reduced by using data from a larger number of sites, but direct multi-source analysis, as outlined here, is preferable.
- The ‘background’ emissions of SO_2, NO_x and NH_3 are among the important parameters determining the external costs from the power sector.
- The current results underline the importance of a site-specific analysis not only for a single power plant, but also for emissions from a whole industrial sector.
- The new insights from the aggregation work might bring up some new questions related to energy policy: What are the optimal and operational instruments for internalisation on a European level, taking into account the spatial variation in damage costs, and the mutual influence of emissions from different industry sectors on the same impact categories?

20.5 References

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21. CONCLUSIONS

This Chapter starts with a review of the major methodological advances made in the ExternE Project in the period 1996 to 1997. Section 21.2 provides a brief review of the applications of this methodology to date. We believe that this demonstrates that the project has been a great success. The section, and the report, then concludes with a review of areas where further research to improve the quality of the results is of most interest from a policy perspective.

21.1 Major Methodological Advances in ExternE, 1996-1997

21.1.1 Assessment of Uncertainty

Our earlier series of reports (European Commission, 1995 a to f) considered the uncertainties of each part of the impact pathway analysis in detail. This developed understanding of the reliability of dispersion models, emission factors, dose response functions, and so on. However, the methodology at that time did not permit these component uncertainties to be integrated in such a way that would provide a properly quantified guide to the reliability of the results for each impact. Significant progress has been reported here, in Chapter 5. In some ways the final part of the current approach, assigning confidence bands to each effect sounds analogous to the methods used previously (based around high, medium, low and confidence bands). However, the use of geometric standard deviations and confidence intervals based on the log normal distribution is a substantial step forward. The earlier system was almost entirely qualitative in comparison.

Some aspects of uncertainty cannot be integrated effectively or transparently within the statistical framework developed in Chapter 5. These typically concern discrete choices on issues such as the approach adopted for valuation of mortality, or discount rate. We believe that these issues are best dealt with using explicit sensitivity analysis.

21.1.2 Assessment of Major Accidents

The valuation of nuclear accidents has been progressed in a number of respects. First it has demonstrated how *ad hoc* rules are not the way forward, and that the existing estimates based on those rules cannot be viewed as valid. Second it has shown that the ‘expected utility framework’ can be applied to estimating risk-adjusted nuclear accident costs, and that the numbers are consistent and reasonable. These can offer an immediate correction to the unadjusted costs. Third it has shown that one can incorporate the differences between public and expert assessment of probabilities into a coherent theoretically sound framework.

21.1.3 Assessment of Global Warming

There are a very large number of possible impacts of climate change most of which will be far reaching in space and time. These have been explored in depth using two models (FUND and the Open Framework). The analysis has accounted for variation in results according to different scenarios for future development in the world to the year 2100. The problems of valuation of impacts against the context of global warming have also been investigated, in particular the difficult, and essentially normative, judgements which need to be made about:

- discount rate
- the treatment of equity,
- the value of statistical life, and
- the magnitudes of higher order effects.

Two ranges of estimates were derived for use in ExternE work. These results are certainly not the final word on global warming damages, but they do help to clarify a number of important issues, in particular with respect to the sensitivities that are likely to be most important. Again, the results of the study represent a significant advance on previously available estimates.

21.1.4 Assessment of Impacts on Human Health

This part of the work has now incorporated epidemiological evidence from a number of European studies, much reducing the previous reliance on the US literature. Assessment of chronic effects of particulates on mortality has been clarified since the earlier report, though further work is needed in this area. Assessment of impacts has been extended to cover a wider range of pollutants, such as heavy metals and some organic carcinogens. The study has also sought to account for variation in the harmfulness of different particulate fractions.

The main development from the perspective of valuation is the adoption of the value of a life year (VOLY) approach, in preference to the value of statistical life (VOSL) in the context of mortality related to air pollution. This accounts for the project team's view that life expectancy should be an important issue in the valuation. However, this issue continues to be the subject of much debate. Considering that there remains a substantial and influential body of opinion remaining in support of the VOSL outside of the project it has been recommended that the VOSL be retained for sensitivity analysis.

21.1.5 Assessment of Ecological Damage

The work presented in this area largely updates information from the earlier report (European Commission, 1995b) on assessment of ecological damages. However, this includes a very substantial increase in the literature reviewed, and an expansion of the number of crops for which exposure-response functions are recommended. The result is that the analysis can now cover most European agricultural production whereas in the past it was limited to under 20% of the total.

We concluded, as before, that it is not currently feasible to integrate damage to ecological systems (beyond productivity of agricultural and forest systems) with other fuel cycle damages using a valuation approach because of a lack of data. Instead consideration has moved to integration of sustainability indicators with the ExternE framework.

21.1.6 Quantification of the Effects of Tropospheric Ozone

A methodology for assessment of the effects of tropospheric ozone is presented, with illustrative values for what is taken to be a representative European case. Further attention is needed to the methodology in this area (see section 21.2.4) though the work reported here has greatly helped in clarifying the issues involved.

21.1.7 Development of the EcoSense Model

The development of the EcoSense model greatly improved consistency of the analysis of air pollution effects through the provision of common models for assessing dispersion and atmospheric chemistry, and common databases of stock at risk. It was used by all of the National Implementation Project teams, allowing cross-comparison of results. Concentration on air pollution in the development of computer tools was justified because of the domination of air pollution damages in the results of the study.

21.1.8 Aggregation of External Costs Data

Two methodologies were developed and assessed. The first was a detailed method, drawing on a multi-source version of the EcoSense model. The second was less sophisticated, but easier to implement, relying on extrapolation of available results. Accuracy of the second method can be improved substantially by expanding the underlying volume of data used for the extrapolation by drawing on a large number of case studies considering systems in different locations.

21.1.9 Consideration of the Externalities of Non-Environmental Externalities

Substantial progress has been made on the methodologies that could be used to incorporate the externalities associated with non-proliferation of nuclear materials and energy security within the ExternE assessment. In the case of energy security, the work indicates that the upper bound on the externality is small in absolute magnitude, i.e. not more than 0.1p/kWh. However, review work suggested that externalities associated with potential nuclear proliferation could be more significant, though quantification of this problem is extremely difficult.

21.2 Areas for Future Work

21.2.1 What is the Demand for Externalities Analysis?

Since the ExternE Project commenced in 1991, eleven main reports have been published under the study (European Commission, 1995 a to f; 1998 a to c; CIEMAT, 1998; IER, 1998), together with numerous others by task teams and national implementation teams. Before addressing the question of whether further work is necessary in any areas; it is sensible to ask whether the work that has been completed to date has been useful. We believe that the answer to this question is unequivocally yes, based on the following list of applications. For the most part these studies have been externally commissioned. To further emphasise the point that the study output is in great demand, it is stressed that the list presented here is not complete.

- Economic evaluation of a draft directive on waste incineration (AEA Technology, IER, ARMINES, Electrowatt, 1996)
- Green Accounting (Metroeconomica and others, 1996)
- Economic evaluation of ambient air quality limits for SO_2, NO_2, fine particles and lead (IVM, NILU and IIASA, 1997)
- External costs and electricity taxation (VITO: see CIEMAT, 1998)
- Introduction of externalities into the electricity dispatch system in Spain (CIEMAT and IIT, see CIEMAT, 1998)
- Incinerators and cars: a comparison of emissions and damages (ARMINES, see CIEMAT, 1998)
- Social costing and the competitiveness of renewable energies (NTUA, see CIEMAT, 1998)
- Solid waste incineration vs. landfilling (IEFE, see CIEMAT, 1998)
- Externalities of energy scenarios in the Netherlands (IVM, see CIEMAT, 1998)
- Cost-benefit analysis of measures to reduce air pollution & decision on building gas fired power plants in Norway (ENCO, see CIEMAT, 1998)
- Strategies for meeting future electricity demand in São Miguel Island (Azores archipelago) (CEEETA, see CIEMAT, 1998)
- Cost benefit analysis of the UNECE Multi-pollutant, Multi-effect Protocol (AEA Technology, Eyre Energy Environment and Metroeconomica, 1998)
- Comparing Costs and Environmental Benefits of Strategies to Combat Acidification in Europe (Krewitt *et al*, 1998)
- Economic evaluation of acidification and ground level ozone (AEA Technology, 1998)
- Development of a simplified methodology for assessment of externalities in developing countries (Spadaro, Markandya, 1998).

ExternE results have also been integrated with energy system modelling exercises, investigating, for example, the effects of carbon and energy taxes set at different levels.

In addition to the above applications, the results generated by ExternE are being extensively used by consultants and others outside of the study team (e.g. IAEA, 1998).

The applications made so far thus include analysis in support of plant technical standards, environmental quality objectives such as air quality limits, and consideration of duty rates. The extensive use of ExternE illustrated here clearly demonstrates that there is much demand for the output of the study. In the interests of consistency the study team believe that it is essential that a closely integrated programme of research should continue in order to keep the methodology at the forefront of analysis of the environmental impacts of energy use. Fragmentation of the study effort will inevitably lead to the development of inconsistency.

21.2.2 Further Development of Tools for Assessment of Transport Externalities

The ExternE Transport Project (IER, 1998) was probably the most significant extension of the project methodology in the period 1996 to 1997. The project clarified many of the issues involved in moving assessment to consideration of mobile sources. However, it did not provide a consistent tool equivalent to the EcoSense model. This should be considered a priority for future work, and is indeed part of a new project being undertaken for EC DGXII. This new programme of work covers rail traffic, aircraft, and shipping, in addition to road transport.

21.2.3 Updating Information on Dose-Response, Valuation, etc.

The project outputs will remain useful only so long as the data held in the system are kept up to date. Given the speed with which this field of research has progressed in recent years it is thus essential that expertise, particularly in health, ecology, valuation and climate change be retained.

21.2.4 Closer Integration of Sophisticated Models for Ozone Assessment

The current assessment of ozone reported in Chapter 16 of this report marks a significant improvement on the earlier ExternE analysis (as presented in European Commission, 1995b, c), which allowed only for very limited local scale modelling. Computer resources needed for this work, and data for the models, were too extensive to permit analysis for more than one or two sites. The revised methodology presented here takes a generalised approach based on the results of the EMEP photochemical model, providing results that are ‘representative’ of the European situation at the present time.

Further refinement is needed, however, because of the extensive spatial variation in ozone at local, regional and global scales. A particular difficulty arises through the non-linearity of ozone formation and loss: an increase in NO_x emissions will lead to elevated O_3 in some parts of Europe, but less in others. The difficulty is increased because of the need to consider future trends in ozone precursor emissions. Within the European Union these will be affected by much legislation in the coming years, such as the development of strategies to deal with acidification and ozone, the Solvents Directive, and the Framework Directive on Ambient Air Quality. Clearly these issues need to be accounted for in order to increase the reliability of the externalities estimates made.

21.2.5 Refinement of Global Warming Assessment

The need to refine global warming assessments will continue for many years, as better input data and more sophisticated modelling approaches become available.

21.2.6 Validation of Modelling Approaches

Further validation activities need to be undertaken to ensure that the results are as accurate as possible, and to assist in the description of uncertainty.

21.2.7 Harmonisation of the Methodology With LCA and the Use of Sustainability Indicators

Better integration of LCA and the use of sustainability indicators will enhance the coverage of the analysis, and promote methodological rigour.

21.2.8 Energy System Optimisation

Some of the case studies undertaken within ExternE (see above) have already addressed questions of energy system optimisation, regarding for example the competitiveness of renewable energy systems. Against the context of the greenhouse gas emission savings to be made as a result of the agreements reached in Kyoto, further consideration needs to be given to this area of analysis. There is a danger that a policy that may appear optimal for addressing one issue (e.g. greenhouse gas abatement) may be sub-optimal when dealing with a series of different issues. The European Commission and UNECE have accepted this in moving towards multiple-pollutant assessments for dealing with acidification, eutrophication and ground level ozone. In the future a similar co-ordinated analysis should be undertaken when revising or setting new air quality limits.

21.2.9 Extrapolation of the Methodology to Developing Countries

There is a pressing need for the methodology developed here to be disseminated more widely, particularly to developing countries. This is made extremely urgent by the rapid pace of development in some areas (e.g. China). Inappropriate choice of technologies now could lead to substantial costs being incurred for many years into the future. There are doubtless arguments about the valuation context in some countries that remain to be addressed. However, we stress that valuation is only one part of the impact pathway analysis: the methodology presented in this report provides numerous other types of result that are relevant to policy making, covering pollutant emissions, occupational health, air pollution impacts and so on.

It will be necessary to ensure in extension of this work to new countries that decisions made as to what constitutes a 'priority impact' are genuinely relevant to the country in question. The same priorities that apply within the European Union may not apply in countries with a different level of environmental legislation in force.

21.3 References

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