ExternE — **Externalities of Energy** — Methodology 2005 Update

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ternalisation of external costs is a key element for the implementation
tainable development in Europe. The so-called "ExternE methodology"
een updated in order to better quantify the social and environmental
ges of energy, especially those provoked by air pollution coming from
y production and consumption. In order to improve the quality of the
nment and to reduce the negative impacts from pollution on human health,
choices have to be made. These choices need to be based on a complete
oherent methodology for accounting for external costs.

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Externalities of Energy

Methodology 2005 Update

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FOREWORD

Externalities are related to social welfare and to the economy. The idea is firstly to measure the damages to society which are not paid for by its main actors; secondly, to translate these damages into a monetary value; and thirdly, to explore how these *external costs* could be charged to the producers and consumers. Indeed, if the market takes into consideration the private costs, policy-makers should try to take account of the external costs.

During the course of the last fifteen years, the European Commission has worked extensively – in particular through socio-economic research in the field of energy - to quantify the energy external costs. The European research allowed a multidisciplinary research team, composed of engineers, economists and epidemiologists, to develop an original methodology, the *Impact Pathway Approach*.

The Impact Pathway Approach tackles issues such as the exposure-response functions; especially health impacts from air pollution, the monetary valuation of these impacts ("value of statistical life"), accidents in the whole energy supply chain, and the assessment of other impacts like global warming, acidification and eutrophication. Models for pollutant dispersion have also been developed and case studies have been performed all around Europe.

Electricity – like transport – is a key factor for economic and social development. Nevertheless, its air pollutants (particles, oxides of nitrogen, sulphur dioxide, etc) provoke damages like morbidity or premature mortality (chronic bronchitis, asthma, heart failure,…). The ExternE research team has made an in-depth analysis of various fuels and technologies in the electricity sector with methodology and results published in 1995 and 1999. An update was necessary to take account of the latest developments both in terms of methods for monetary valuation and technological development.

The ExternE methodology is widely accepted by the scientific community and is considered as the world reference in the field. With ExternE, and this new "green accounting framework", a ranking of technologies can be made according to their social and environmental impacts. Internalising external costs, by taxing the most damaging technologies or by subsidising the cleanest and healthiest ones, can give an impetus to new technologies and could help to achieve a more sustainable world.

Achilleas Mitsos
Director-General for Research

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In addition, this work draws on the contributions and results of numerous colleagues, who have contributed to the development of the methodology in earlier projects of the ExternE project series.

A list of the core teams involved in the development of the ExternE methodology (together with further documentation and results) can be found at www.externe.info.

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1 Executive Summary

1.1 Overview of the Methodology

The ExternE methodology provides a framework for transforming impacts that are expressed in different units into a common unit – monetary values. It has the following principal stages:

1) Definition of the activity to be assessed and the background scenario where the activity is embedded. Definition of the important impact categories and externalities.
2) Estimation of the impacts or effects of the activity (in physical units). In general, the impacts allocated to the activity are the difference between the impacts of the scenario with and the scenario without the activity.
3) Monetisation of the impacts, leading to external costs.
4) Assessment of uncertainties, sensitivity analysis.
5) Analysis of the results, drawing of conclusions.

The ExternE methodology aims to cover all relevant (i.e. not negligible) external effects. However, in the current state of knowledge, there are still gaps and uncertainties. The purpose of ongoing research is to cover more effects and thus reduce gaps and in addition refine the methodology to reduce uncertainties. Currently, the following impact categories are included in the methodology and described in detail in this report:

1) Environmental impacts:

Impacts that are caused by releasing either substances (e.g. fine particles) or energy (noise, radiation, heat) into the environmental media: air, soil and water. The methodology used here is the impact pathway approach, which is described in detail in this report.

2) Global warming impacts:

For global warming, two approaches are followed. First, the quantifiable damage is estimated. However, due to large uncertainties and possible gaps, an avoidance cost approach is used as the recommended methodology.

3) Accidents:

Accidents are rare unwanted events in contrast to normal operation. A distinction can be made between impacts to the public and occupational accident risks. Public risks can in principle be assessed by describing the possible accidents, calculating the damage and by multiplying the damage with the probability of the accidents. An issue not yet accounted for here is the valuation so-called 'Damocles' risks, for which high impacts with low probability are seen as more problematic than vice versa, even if the expected value is the same. A method for addressing this risk type has still to be developed.

1.2 The Impact Pathway Approach

The impact pathway approach (IPA) is used to quantify environmental impacts as defined above. As illustrated in Figure 1.1, the principal steps can be grouped as follows:

- Emission: specification of the relevant technologies and pollutants, e.g. kg of oxides of nitrogen (NO_x) per GWh emitted by a power plant at a specific site;
- Dispersion: calculation of increased pollutant concentrations in all affected regions, e.g. incremental concentration of ozone, using models of atmospheric dispersion and chemistry for ozone (O_3) formation due to NO_x ;
- Impact: calculation of the cumulated exposure from the increased concentration, followed by calculation of impacts (damage in physical units) from this exposure using an exposure-response function, e.g. cases of asthma due to this increase in O_3;
- Cost: valuation of these impacts in monetary terms, e.g. multiplication by the monetary value of a case of asthma.

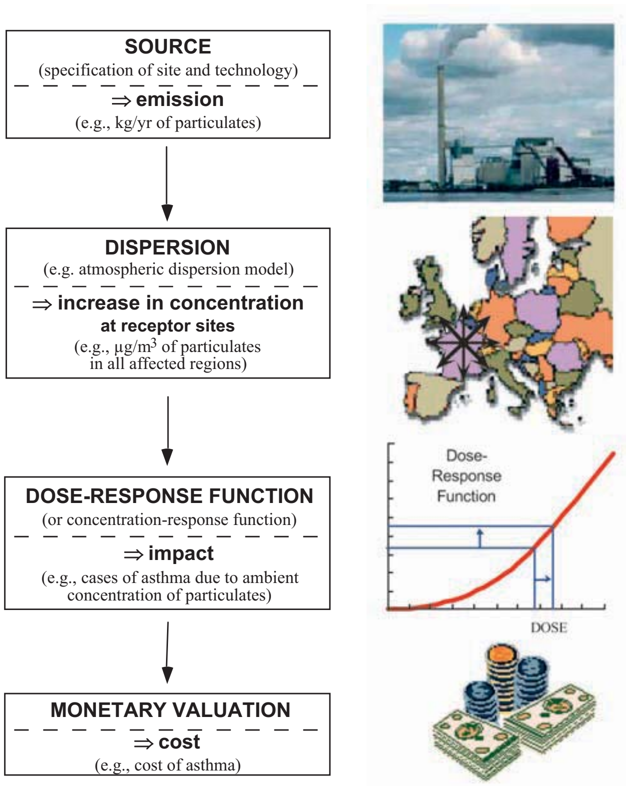

Figure 1.1 The principal steps of an impact pathway analysis, for the example of air pollution.

Whereas only the inhalation dose matters for the classical air pollutants (PM_{10}, NO_x, SO_2 and O_3), toxic metals and persistent organic pollutants also affect us through food and drink. For these a much more complex IPA is required to calculate ingestion doses. Two models were developed for the assessment of external costs due to the emission of the most toxic metals (As, Cd, Cr, Hg, Ni and Pb), as well as certain organic pollutants, in particular dioxins.

Table 1.1 Air pollutants and their effects on health.

Primary Pollutants	Secondary Pollutants	Impacts
Particles (PM_{10}, $PM_{2.5}$, black smoke)		mortality cardio-pulmonary morbidity (cerebrovascular hospital admissions, congestive heart failure, chronic bronchitis, chronic cough in children, lower respiratory symptoms, cough in asthmatics)
SO_2		mortality cardio-pulmonary morbidity (hospitalisation, consultation of doctor, asthma, sick leave, restricted activity)
SO_2	Sulphates	like particles?
NO_x		morbidity?
NO_x	Nitrates	like particles?
NO_x+VOC	Ozone	mortality morbidity (respiratory hospital admissions, restricted activity days, asthma attacks, symptom days)
CO		mortality (congestive heart failure) morbidity (cardio-vascular)
PAH diesel soot, benzene, 1,3-butadiene, dioxins		cancers
As, Cd, Cr-VI, Ni		cancers other morbidity
Hg, Pb		morbidity (neurotoxic)

In terms of costs, health impacts contribute the largest part of the damage estimates of ExternE. A consensus has been emerging among public health experts that air pollution, even at current ambient levels, aggravates morbidity (especially respiratory and cardiovascular diseases) and leads to premature mortality (see Table 1.1). There is less certainty about specific causes, but most recent studies have identified fine particles as a prime culprit; ozone has also been implicated directly. The most important cost comes from chronic mortality due to particles (this term, chosen by analogy with acute and chronic morbidity impacts, indicates that the total or long-term effects of pollution on mortality have been included, in contrast to acute mortality impacts, which are observed within a few days of exposure to pollution).

1.3 Methods for Monetary Valuation

The impact pathway requires an estimation of the impacts in physical terms and then a valuation of these impacts based on the preferences of the individuals affected. This approach has been successfully applied to human health impacts, for example, but in other areas it cannot be fully applied because data on valuation is missing (e.g. acidification and eutrophication of ecosystems) or because estimation of all physical impacts is limited (e.g. global warming).

For these cases, a second best approach is better than having no data. Therefore the use of approaches that elicit implicit values in policy decisions to monetise the impacts of acidification and eutrophication and of global warming has been explored. Table 1.2 gives a general overview of the methods for quantifying and valuing impacts.

Table 1.2 Overview of methods used in ExternE to quantify and value impacts.

	Air pollution			Global warming
	Public health	Agriculture, building materials	Ecosystems	
ExternE, "Classical" impact pathway approach				
Quantification of impacts	Yes	Yes	Yes, critical loads	Yes, partial
Valuation	Willingness to pay (WTP)	market prices		Yes, WTP & market prices
Extension: Valuation based on preferences revealed in				
Political negotiations			UN-ECE; NEC	Implementing Kyoto, EU
Public referenda				Swiss Referenda

Under certain assumptions the costs of achieving the well-specified targets for acidification, eutrophication and global warming can be used to develop shadow prices for pollutants or specific impacts from pollutants. These shadow prices can be used to reflect these effects for comparison of technologies and fuel cycles.

For global warming damage cost estimates of ca. €9/tCO_2 were derived for a medium discount rate. However, this figure is conservative in the sense that only damage that can be estimated with a reasonable certainty is included; for instance impacts such as extended floods and more frequent hurricanes with higher energy density are not taken into account, as there is not enough information about the possible relationship between global warming and these impacts.

Thus, to account for the precautionary principle, we propose to use an avoidance costs approach for the central value. The avoidance costs for reaching the broadly accepted Kyoto target is roughly between €5 and €20 per t of CO_2. In addition it is now possible to analyse the prices of the tradeable CO_2 permits, which increased from end of July 2005 to the beginning of October 2005 from about €18/tCO_2 to about €24/tCO_2. This confirms the use of €19/t CO_2 as a central value. The lower bound is determined by the damage cost approach to about €9/t CO_2.

More stringent reduction targets, e.g. the EU target of limiting global warming to 2°C above pre-industrial temperatures may lead to marginal abatement costs as high as \$350/tC = ca. €95/t CO_2. However it is still an open question whether such an ambitious goal with such high costs will be accepted by the general population. Thus, as an intermediate target, the Dutch value of ca. €50/t CO_2 could be used as an upper bound for sensitivity analysis.

In the context of acidification and eutrophication the study shows that a simple analysis may not be correct, i.e. abatement costs for SO_2 and NO_X need to be corrected for other impacts. By analysing the decisions of policy makers in detail, shadow prices for exceedance of critical loads for eutrophication and acidification (ca. €100 per hectare of exceeded area and year with a range of €60 - 350/ha year) have been derived.

1.4 Uncertainties

Damage cost estimates are notorious for their large uncertainties and many people have questioned the usefulness of damage costs. The first reply to this critique is that even an uncertainty by a factor of three is better than infinite uncertainty. Second, in many cases the benefits are either so much larger or so much smaller than the costs that the implication for a decision is clear even in the face of uncertainty. Third, if policy decisions are made without a significant bias in favour of either costs or benefits, some of the resulting decisions will err on the side of costs, others on the side of benefits. Analyses of the consequences of such unbiased errors found a very reassuring result: the extra social cost incurred because of uncertain damage costs (compared to the minimal social cost that one would incur with perfect knowledge) is remarkably small, less than 10 to 20% in most cases even if the damage costs are in error by a factor three. However, without any knowledge of the damage costs, the extra social cost could be very large.

One possibility to explore the uncertainties in the context of specific decisions is to carry out sensitivity analyses and to check whether the decision (e.g. implementation of technology A instead of technology B) changes for different assumptions (e.g. discount rate, costs per tonne of CO_2, valuation of life expectancy loss). It is remarkable that certain conclusions or choices are robust, i.e. do not change over the whole range of possible values of external costs. Furthermore, it can be shown that the ranking of electricity production technologies, for example, with respect to external costs does not change if assumptions are varied. A further option is to explore how much key values have to be modified before conclusions change. It can then be discussed whether the values triggering the change in decision can be considered realistic or probable.

A considerable share of uncertainties is not of a scientific nature (data and model uncertainty) but results from ethical choices (e.g. valuation of lost life years in different regions of the world) and uncertainty about the future. One approach to reduce the range of results arising from different assumptions on discount rates, valuation of mortality, etc. is to reach agreement on (ranges of) key values. Such "conventions for evaluating external costs", resulting from discussion of the underlying issues with relevant social groups or policy makers, help in narrowing the range of costs obtained in sensitivity analyses. This would help to make decision making in concrete situations easier and to focus on the remaining key issues to be solved in a specific situation.

2 Introduction

For almost 15 years the European Commission has supported the development and application of a framework for assessing external costs of energy use. In the ExternE (Externalities of Energy) project series, the impact pathway methodology has been developed, improved and applied for calculating externalities from electricity and heat production as well as transport.

The ExternE Project commenced in 1991 as the European part of a collaboration with the US Department of Energy in the 'EC/US Fuel Cycles Study'. Successful collaboration at that time produced a workable methodology for detailed quantification of the external costs of fuel cycles. A series of reports were published in the USA and in Europe in 1994 and 1995, the European reports covering:

- Volume 1: Summary
- Volume 2: Methodology
- Volume 3: Coal and Lignite
- Volume 4: Oil and Gas
- Volume 5: Nuclear
- Volume 6: Wind and Hydro

Continued funding allowed the European study team to expand in the next phase of the study which ran from 1996 to 1997, bringing in additional expertise and broadening the geographical coverage of the study within the European Union. By this time all EU Member states except Luxembourg were included, and from outside the EU, Norway. At the end of that phase, four further reports were produced by the European team:

- Volume 7: Methodology Update 1998
- Volume 8: Global Warming Damages
- Volume 9: Waste, PV, New Power Technologies and End Use Technologies
- Volume 10: National Implementation Results

Since then the methodology was taken further in a number of projects. The impact pathway analysis was extended to the environmental media soil and water. New scientific knowledge was included, above all in the areas of health impact quantification, modelling of global warming effects, and monetary valuation. Contingent valuation studies on the valuation of changes in life expectancy were carried out. Furthermore monetary values were derived based on preferences revealed in political negotiations. These can be used where the impact pathway approach cannot be fully applied due to missing knowledge.

As is evident from its title, this report supersedes the original Volumes 2 and 7 on methodology. Elements of the earlier reports have been retained where appropriate. However, this report mainly consists of new or revised text, reflecting the significant advances made to the end of 2004.

3 Purpose and General Methodology

3.1 Purpose of Quantifying External Costs

In this report the methodology for calculating external costs used in ExternE is explained. This evokes the question, what external costs are and for what purposes they can be used.

Human activities cause damages and impose risks on human beings, ecosystems and materials. For instance, a power plant when producing electricity may emit pollutants that are transported in the atmosphere and then when inhaled can create a health risk or after deposition can disturb ecosystems. The power plant operator has no incentive to account for this damage, when making decisions His duty is to respect the emission thresholds imposed by environmental regulation, but not the avoidance of further small risks and damages. The damages occurring thus are external effects, i.e. not taken into account by the person or institution causing the effects. In order to be able to assess and compare the external effects with each other and with costs, it is advantageous to transform them into a common unit; the choice of a monetary unit here has advantages described later. Thus converting external effects into monetary units results in external costs. These external costs are not accounted for by the decision maker; thus they should be internalised by using appropriate instruments, e.g. a tax.
Thus, *an external cost arises, when the social or economic activities of one group of persons have an impact on another group and when that impact is not fully accounted, or compensated for, by the first group.*

Why would we want to calculate external costs and for what purposes do we need or use them? There are a number of purposes, which are described in the following.

When **investment decisions** are made, e.g. about which power plant technology to use or where to site a power plant, it is evident that it would be of interest for society to take environmental and health impacts into account and include the external effects into the decision process, i.e. to internalise external costs. Of course, before internalisation the external costs have to be estimated. More precisely, marginal external costs are needed, i.e. the additional external costs that arise when the investment alternative is implemented. This implies that not only external costs occurring during operation, but also during construction, provision of energy carriers and materials, waste disposal, dismantling, etc., i.e. the full life cycle, have to be accounted for. To support the decision process, the social costs of the investment alternatives, i.e. the sum of internal and external costs, can then be compared. If decisions are to be taken now, but the consequences of the decisions reach decades into the future, the possible future costs have to be estimated.

In a similar way external cost estimates are useful for carrying out **technology assessments**, and thus to find out the principal weaknesses and strengths of a technology and to be able to assess the overall performance and usefulness of a technology; this would for example help to answer questions about whether and where the technology would need further improvement, and whether subsidising it or supporting further research might be justified. As not necessarily specific technologies at specific sites are analysed, future typical or average marginal external costs for typical technologies at different sites would be needed.

Not only investments cause external costs, but also **consumption of consumer goods, whereby the choice between alternative technologies or consumer goods can influence the size of externalities considerably**. Again, marginal external costs are needed, for example the costs that are caused by driving a car on a certain road or type of road or the costs of using a stove for heating. The best way of internalising these costs is via imposing taxes that are equal to the external costs, so that prices reflect the true costs and tell the ecological truth. However, as it is often not feasible to fix a different tax for each individual case, averaged external costs for classes of goods, sites or activities are used to determine the tax to be imposed.

A fourth very important field of application is the performance of **cost-benefit-analyses for policies and measures that reduce environmental and health impacts**. Policies and measures for reducing environmental pollution generally imply additional costs for industry and consumers. Thus it is important for the acceptance of the measure to show that the benefits, for example reduced health risks, outweigh or justify the costs. The benefit can be expressed as avoided external costs. To calculate the avoided external costs, it is necessary to create two scenarios: a baseline scenario, which describes a development without the implementation of the measure or policy and a scenario including it. Then the impacts occurring for the two scenarios are calculated. The difference of the impacts is monetised; this gives the avoided external costs or benefits (provided that the impacts of the scenario with the measure are lower than for the baseline scenario). These benefits can then be compared with the costs. If benefits are larger than costs, the policy or measure is beneficial for society's welfare.

The fifth area of application is the assessment of health and environmental impacts occurring in a region due to activities of different economic branches, in short **green accounting**. For example one could monetise the health effects occurring due to emission of different pollutants, and can then rank different source categories, economic sectors or pollutants according to their health impacts, compare health effects in different countries or imposed from one country to another or compare health effects of different years to find out whether the situation is improving. Again, the external costs of two scenarios are compared, a baseline scenario and a scenario where the activities that should be assessed are omitted. The difference is then allocated to these activities.

3.2 Basic Principles of the Methodology

The applications mentioned above all have something in common: having calculated different impacts (risks, damage) or indicators, that indicate to what extent objectives are fulfilled, it is necessary to compare these impacts with each other and with costs. To compare technologies or assess policies, it has to be found out whether one composition of impacts and costs is better or worse than another composition. This is not straightforward, as the different impacts have different units, so they cannot be added directly. So, before being able to add them, it is necessary to transform them into a common unit. The ExternE methodology provides a framework for doing this. The basic principles of the methodology are derived in the following.

1) All the applications mentioned above imply that there are different effects and impacts that somehow have to be weighted with each other to get an overall assessment of whether one basket of impacts is better or worse than another. The first principle of the ExternE methodology is that this **assessment or weighting of impacts is as far as possible carried out using quantitative figures and procedures.** The reason is that only quantitative algorithms ensure the necessary transparency and reproducibility of results.
2) Secondly, **the common unit into which impacts are transformed is a monetary unit**. This has a number of advantages. First, units are conceivable. The importance of an impact in monetary units, say €10,000, can be directly and intuitively grasped, as one can compare it with the utility of the goods and services that one could buy with this amount. Whereas an amount of say 120 utility points does not say anything about the importance of the impact. Secondly, monetary values are transferable from one application to another. This is because monetary units are defined independent of the assessment process. So if a monetary valuation of the risk to get a certain disease, e.g. bronchitis, has been found, this value can – with some caution and adjustment – then be used in a further analysis, where this disease occurs, without having to carry out a new survey on its monetary value. Thirdly, the above mentioned applications at some stage require the use of monetary units. So in order to compare costs with benefits, it is necessary to convert benefits into monetary units. It would of course also be possible to convert costs into some benefit unit like ecopoints, but this is obviously less useful due to the first reason mentioned. For internalising external effects with taxes, it is also obviously necessary to express these effects in monetary units.
3) How is it possible to get a measure for the relative importance of impacts and thus for the weighting factors or algorithms needed? As no natural law exists that somehow weighs impacts with different units, the logical possibility is to measure the preferences of the population. This can be done with a number of methods, e.g. by asking for or observing the willingness to pay to avoid a certain impact. The only alternative would be to measure the preferences of elected representatives of

the population, with the argument that representatives are or can be better informed than the public. However, these representatives change, so that benefit transfer is difficult or not possible. Furthermore experience with multiattribute utility analysis shows that decision makers are often not willing to expose their preference structure, possibly because they fear that they lose influence on their decisions. Although in ExternE it is also possible to use revealed preferences of decision makers for example, the preferred way is to directly measure preferences of the population. To get useful results, impacts should be described and explained as well as possible before measuring preferences. Given that it would cost too much effort to ask the whole population, it seems sufficient to ask a representative sample of the population. Thus, the **assessment of impacts is based on the (measured) preferences of the affected well-informed population.**

4) To be able to get meaningful results, the interviewed persons have to understand the change of utility that occurs due to the impact to be assessed. This implies that **it is important to value a damage, not a pressure or effect**. For instance, it is not useful to ask for the willingness to pay to avoid an amount of emissions, say 5 tonnes of NO_x, as no one – at least without further information or knowledge – can judge the severity of this or the damage or loss of utility caused by this emission. On the other hand, if somebody is asked for an assessment of a concrete health risk, e.g. a cough day, he can compare this impact with other impacts and changes of utility that he experiences.
5) An important aspect is that external costs depend on the time and site of the pressure. For instance, if emissions of air pollutants occur in a densely populated area, the health of more people is at risk than for a site where equal amounts of pollutants are emitted but in a less densely populated area. The emissions of sulphur dioxide are more harmful in areas where ammonia concentrations in the atmosphere are higher, because then more ammonium sulphate is formed, which causes more health damage than SO_2. Noise in a city at night is more annoying than a similar noise level outside the city during the day. **The methodology should thus be capable of calculating site and time dependent external costs.** Only a detailed bottom-up calculation allows a close appreciation of such site, time and technology dependence. Thus for most environmental impacts **the so-called 'Impact Pathway Approach' is used**, that follows the complete chain of causal relationships, starting with the emission of a burden through its diffusion and conversion in the environmental media to its impact on the various receptors and finally the monetary valuation of its impacts.
6) Depending on the nature of the policy question, **average or aggregated external costs can then be calculated** as needed to support the implementation of different policy instruments.

The methodology thus has the following principal stages:

1) Definition of the activity to be assessed and the background scenario where the activity is embedded. Definition of the important impact categories and externalities.
2) Estimation of the impacts or effects of the activity (in physical units). In general, the impacts allocated to the activity are the difference between the impacts of the scenario with and the scenario without the activity.
3) Monetisation of the impacts, leading to external costs.
4) Assessment of uncertainties, sensitivity analysis.
5) Analysis of the results, drawing of conclusions.

The basic elements of the methodology are:

- External effects: External effects arise if, due to the activities of one person or group of persons, an impact on another group occurs that is not taken into account or compensated for by the first group. The impact has to have an influence on the utility or welfare of the second group. External effects can be positive or negative. Further distinction can be made between direct use values (direct effects on the utility of the persons whose preference is measured), indirect use values (effects on the utility of persons, e.g. children, other than those whose preference is measured) and non-use or existence values.
- Indicators: Indicators are used to express the amount of the external effect in a quantitative way. If, for instance, the effect is a change of the risk to get chronic bronchitis, the indicator might be the change in the number of cases of chronic bronchitis per 100 000 inhabitants.
- Functions for monetary valuation: A function that transfers the indicator values into monetary values. If the relation between indicator and monetary value is linear, a parameter MV per unit of indicator is given. Methods to derive this function or parameter are described later in this chapter.

The ExternE methodology aims to cover all relevant (i.e. not negligible) external effects. However, in the current state of knowledge, there are still gaps and uncertainties. The purpose of ongoing research is to cover more effects and thus reduce gaps and in addition refine the methodology to reduce uncertainties. Currently, the following impact categories are included in the methodology and thus described in detail in the following chapters:

1) Environmental impacts:

Environmental impacts here mean impacts that are caused by releasing either substances (e.g. fine particles) or energy (noise, radiation, heat) into the environmental media: air, soil and water. The substances and energy are transported and transformed and finally reach receptors (humans, plants, materials, ecosystems), where they cause risks and damage. Clearly, the methodology to use here is the impact pathway approach. Due to lack of knowledge, the pathway from emission to damage can sometimes not be quantified; in that case, other second best methods, e.g. marginal avoidance costs or restoration costs are used.

2) Global warming impacts:
For global warming, two approaches are followed. First, the quantifiable damage is estimated based on a top-down approach; i.e. the total damage of a scenario is calculated and then distributed on the emissions of greenhouse gases. However, due to large uncertainties and possible gaps, an avoidance cost approach is used in addition. This means that the marginal avoidance costs to reach given emission reduction targets are used.

3) Accidents:
Accidents are rare unwanted events in contrast to normal operation. A distinction can be made between impacts to the public and occupational accident risks. Public risks can in principle be assessed by describing the possible accidents, calculating the damage and by multiplying the damage with the probability of the accidents. An issue not yet accounted for here is risk aversion, which means that high impacts with low probability are seen as more problematic than vice versa, even if the risk is the same. For occupational risks statistics are usually available; the difficult item here is to judge to what extent these risks are external.

4) Energy security:
If unforeseen changes in availability and prices of energy carriers occur, this has impacts, for instance on economic growth. A first attempt to estimate the order of magnitude of the resulting external costs has been made in the project 'ExternE-POL', however the methodology is currently revised within the project 'CASES' and will be described in a revised version of this report as soon as available.

In addition, there are a number of issues that are sometimes seen as important for the decision process, but are – at least according to the opinion of the ExternE team – not external costs. These include:

- **Impacts on employment:**
  Employment is influenced by the labour market; thus impacts on employment are not, according to economic theory, external costs as defined above. However, they nevertheless are usually an important argument in any investment decision.
  We first have to note that there are direct effects, i.e. the construction and operation of a power plant would of course lead to the creation of working places. On the other hand, there are indirect effects, e.g. the operation of the plant might lead to a change in electricity prices, which changes the costs for producing other goods and thus the demand for these goods and so on.
  In general it is more the change of the distribution of working places that might have an important local effect. However, these effects are currently not included in

ExternE and thus have to be taken into account separately within the decision making process.

- **Depletion of non-renewable resources:**
  According to Hotelling's theory the depletion of exhaustible resources is considered in the prices of the resources, thus costs of depletion are internal. However, if one assumes that the current interest rates are higher than the social preference rate that should be used for social issues, then some adjustment should be made. However, this is not yet considered within ExternE.

By far the most important of the external effects are the environmental impacts. Thus, the following chapters explain the impact pathway approach used for these impacts in detail.

Which marginal costs should be estimated and internalised?

If marginal external costs are broadly internalised, i.e. in one or several sectors of an economy, this might lead to decisions that in turn lead to a large change of emissions. Now, due to this change in emissions, marginal external costs may change, i.e. the damage caused by one unit of pollutant may change, because it might depend on the overall level of emissions and thus concentrations of pollutants. Three cases have to be distinguished:
First: if marginal costs do not depend on the emission level, e.g. as the concentration-response-relationship is linear and no chemical conversion of substances occurs, marginal external costs stay constant.

Second, if marginal costs decrease with decreasing emission levels, after the internalisation the emission levels and marginal costs decrease; thus, if external costs from the starting point are internalised, this would result in a too far-reaching reduction of emissions. Two possibilities to avoid this problem are available. First, one could adjust the internalised marginal costs according to the progress achieved in reducing emissions. However, this might result in investments in emission reduction measures that are not efficient. So the better possibility is to estimate the marginal external costs that occur at the optimal point, i.e. where marginal external costs are equal to marginal avoidance costs. This of course can only be calculated if the avoidance cost curve is available, which involves some uncertainties.

Third, if marginal costs increase with decreasing emission levels, the internalisation at current levels leads to an emission reduction that is too low. There are two examples where this occurs:
The first is the case of tropospheric ozone. In certain regions, a limited reduction of NO emissions might not lead to a significant reduction of ozone, in urban areas ozone

concentrations might even increase. Only after NO emissions are substantially reduced, one might achieve a shift from a VOC limited regime to a NO limited one.
The second example is noise. In a very busy street, an additional vehicle will increase the noise level much less than in a quiet street.

It is clear that, in both cases, the aim of an internalisation to initiate measures to reach the optimal level of pollution control is only achieved if not the marginal costs before the internalisation but rather at the optimal level are internalised. This has to be estimated based on scenario calculations and reduction costs.

3.3 Methods for Monetisation

The following sections discuss the use of non-market valuation techniques for end-points of dose-response functions and alternative approaches for monetary valuation where no reliable impact estimates are possible.

3.3.1 Non-market valuation techniques for end-points of dose-response functions

Non-market valuation is a technical term used to describe the idea that a number of welfare components in the valuation of external costs or any project appraisals do not have the value of that welfare expressed in a market price. For example, environmental goods and services generally have characteristics[1] that make it difficult or even impossible for markets in these services to function well. The public good feature of environmental services leads to market failure in a sense that individuals are not free to vary independently the level of the services they consume (Freeman, 2003). Thus, non-market valuation techniques are necessary to estimate monetary values of welfare changes in consumption of environmental services. Other examples include the welfare effects on health of changes in pollution. In neither case is the good or service traded in a market but it is recognised that there is a welfare change. In order to represent these types of welfare changes, we have to adopt non-market valuation techniques to measure the size of the welfare changes.

Generally non-market valuation methods are classified according to the origin or source of the data analysed. Mitchel and Carson (1989) observe that data on environmental use often come from either observations of individuals acting in real-world settings or from individuals' responses to hypothetical questions that aim to elicit individuals' preferences in regard to the environmental good or service. The valuation methods based on the former type of data source are called **revealed preference** methods, while the methods based on the latter are known as **stated preference** methods (Freeman, 2003)[2].

[1] Non-excludability and non-rivalry in consumption are typical characteristics of environmental services, such as air quality and noise. These are also characteristics of public goods.
[2] This section presents a general overview of the non-market valuation methods. For a formal definition

Using revealed preference methods, also known as behavioural methods, the researcher observes individual behaviour towards a market good with connection to the non-market good or service investigated, assuming individuals' behaviour reflects utility maximisation subject to income constraint. From this behaviour the analyst infers the value individuals pose on the non-market good or service of interest. For example, analysts can use individuals' behaviour in the house market to estimate the value of changes in air quality (non-market service of interest), which is an important attribute of the marketed good (houses).

Revealed preference techniques can be divided into direct and indirect methods (Navrud, 2004). **Direct revealed preference methods** include simulated market exercises, i.e., constructing a real market for a non-market good. An example of a revealed preference direct method is based on observed choices in a **referendum** exercise, where individuals are offered a fixed quantity of a good at a given price on a 'yes-no' basis. Individuals' choices reveal if the value of the offered good is greater or less than individuals' maximum willingness to pay for the offered good. In order to use the results of this type of referendum exercise to value a good, data on voting behaviour is needed for different levels of the good at a fixed price or for a fixed level of the good at different prices. However, in most referendum exercises the voters only vote for or against one specified price for the provision of one level of the good. Contingent valuation surveys, discussed below, overcome this problem by simulating referendum exercises at different levels of prices and good provision[3].

Table 3.1 summarises the non-market valuation techniques and their classification.

Table 3.1 Classification of non-market valuation techniques.

<table>
<tr><td></td><td>Indirect</td><td>Direct</td></tr>
<tr><td rowspan="4">Revealed preferences</td><td rowspan="3">Household production function approach
• Travel cost method
• Averting costs</td><td>Simulated markets</td></tr>
<tr><td>Actual referenda</td></tr>
<tr><td>Market prices</td></tr>
<tr><td>Hedonic price analysis</td><td>Replacement costs</td></tr>
<tr><td>Stated preferences</td><td>Choice experiments
• Conjoint analysis
• Contingent ranking
• Contingent rating
• Pair wise comparisons</td><td>Contingent valuation</td></tr>
</table>

Source: Adapted from Navrud (2004) and Freeman (2003).

of these valuation methods refer to Freeman (2003).

[3] Navrud (2004) argues that another advantage of contingent valuation surveys over referendum exercises is that they secure a more representative sample of the population than a referendum (actual referenda), "which often have low participation rates and are dominated by better-educated and better-off citizens" (Navrud, 2004).

Current thinking regarding the pros and cons of the different individual techniques are outlined in the sub-section below.

3.3.2 Description of techniques used

Direct techniques

Direct techniques include using market prices and replacement costs. **Market prices** are used when there is a physical impact – through a dose-response or exposure-response function – on the production function of a given market good. In this case, the physical impact is multiplied by the market price of the affected good to estimate an economic (use) value of the non-market good. As an example, the impact of air pollution from electricity generation or transport (non-market good) on crops (market goods) can be cited. If the crop damage is small enough to avoid changes in relative market prices, in which case changes in consumer and producer surpluses have to be taken into account, then the reduction in crop output can be multiplied by the crops' market price to estimate the impact of air pollution in crop damage. The great advantage of this method is that it relies on the use of market prices to derive values rather than having to infer values through indirect means.

The **replacement or restoration cost** method assumes that the economic cost of a non-market good can be estimated by the market price of a substitute market good that can replace or restore the original quantity or quality level of the non-market good. Navrud (2004) cites that it has been used to estimate economic damages from soil erosion by using market prices for soil and fertilisers to calculate what it would cost to replace the lost soils, and also to calculate loss of ecosystem functions. The author argues that this method estimates arbitrary values that might bear little relationship to true social values – e.g. individuals' willingness to pay for the restoration of environmental and cultural amenities may be more or less than the cost of replacement. Nevertheless, their advantage is seen as being that they make direct use of market prices.

Indirect revealed preference techniques

These techniques use models of relationships between market goods and the non-market good of interest, assuming that there is some kind of substitute or complementary relationship between both goods. Examples of these methods include the household production model together with the travel cost method, the averting behaviour method and hedonic price analysis**.**

The **household production function** approach investigates changes in consumption of commodities that are substitutes or complements for the non-market good. The **travel cost** method estimates recreational use values through the analysis of travel expenditures incurred by consumers to enjoy recreational activities The travel expenditures, participation rates, visitor attributes and information about substitute sites can be used to infer the demand for recreation and the consumer surplus as the welfare measure associated with changes in the environmental attributes of the recreational site. The travel cost model is based on the recognition that the cost of travelling to a site is an important component of the full cost of a visit and that there would be variation in travel costs across any sample of visitors (Freeman, 2003). Travel cost models are very sensitive to many aspects including to model specification, the choice of functional forms, treatment of travel time and substitute sites. The quality of estimates generated by travel cost models depends on how the analyst deals with those issues. However, travel cost models have the advantage of being relatively cheap to perform when compared to standard preference methods (described below).

Averting costs, or defensive/preventive expenditures, assumes that individuals spend money on certain activities that reduce their risks (e.g. impact of pollution, risks of accidents) and that these activities are pursued to the point where their marginal cost equals their marginal value of reduced impact. Averting goods related to pollution include air filters, water purifiers and noise insulation, while averting goods that reduce risks of death may include seat belts and fire detectors. One criticism of the averting cost method is that the consumer decides whether or not to buy the averting good depending on whether his or her marginal benefit is not less than the marginal cost of purchasing the good. The marginal cost equals the marginal benefit only for the last person to purchase the averting good; for all other consumers, the willingness to pay exceeds the marginal cost of a reduction in risks/impacts. Another problem arises when the averting activity produces joint benefits, such as when it reduces the risk of injury or property damage as well as the risk of death. As with the replacement cost method, however, the advantage of the technique is seen as being that it makes direct use of market prices.

Hedonic price analysis refers to the estimation of implicit prices for individual attributes of a market commodity when an environmental good or service can be viewed as attributes of a market commodity, such as properties or wages. The hedonic price model provides the basis for deriving welfare measures from observed differences in property prices or wages offered in the job market. The method is based on the assumption that house characteristics (job characteristics) yielding differences in attributes across houses (jobs) should be reflected in property value (wage) differentials. Thus, just as wages are higher in risky occupations to compensate workers for their increased risks, property values may be lower in polluted areas to compensate residents for their increased risks. The property market is then used to infer the willingness to pay to reduce risks or disutility, through a hedonic price function. However, the hedonic price function is sensitive to the specification and functional form and a number of econometric issues are generally involved in the estimation of the value of the desired attribute. It is also a resource intensive exercise to make these estimates.

Stated preference is a generic name for a variety of techniques including the contingent valuation and choice experiments like contingent ranking, contingent choice and conjoint analysis. In the stated preference approach, researchers pose contingent or hypothetical questions to respondents, inducing responses that trade-off improvements in public goods and services for money. From the responses, preferences for the hypothetical good or the value of changes in provision of the hypothetical good can be inferred. The hypothetical nature of stated preference is at the same time one major advantage in regard to other approaches[4] and, on the other hand, the main argument against stated preference methods.

Contingent valuation is a survey method in which respondents are asked to state their preferences in hypothetical or contingent markets, allowing analysts to estimate demands for goods or services that are not traded in markets. In general, the survey draws on a sample of individuals who are asked to imagine that there is a market where they can buy the good or service evaluated. Individuals state their individual willingness to pay for a change in the provision of the good or service, or their minimum compensation (willingness to accept) if the change is not carried out. Socio-economic characteristics of the respondents – gender, age, income, education etc. – and demographic information are obtained as well. If it can be shown that individuals' preferences are not stated randomly, but instead vary systematically and are conditioned by some observable demographic characteristics, then population information can be used to forecast the aggregate willingness to pay for the good or service evaluated. The contingent valuation method has been widely used for estimating environmental benefits in particular.

The literature on the contingent valuation method's advantages and disadvantages is large (e.g. Mitchell and Carson, 1989; Bateman *et al.*, 2002). A key problem to resolve in a contingent valuation study is to make the scenario sufficiently understandable, clear and meaningful to the respondent, who must understand clearly the changes in characteristics of the good or service he or she is being asked to value. The mechanism for providing the good or service must also seem plausible in order to avoid scepticism that the good or service will be provided, or the changes in characteristics will occur. However, perhaps the most serious problem related to contingent valuation studies may be the fact that the method provides hypothetical answers to hypothetical questions, which means no real payment is undertaken. This fact may induce the respondent to overlook his or her budget constraint, consequently overestimating his or her stated willingness to pay. Another criticism refers to the fact that researchers cannot know for sure that individuals would behave in the same way in a real situation as they do in a hypothetical exercise.

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[4] For example, the hypothetical nature of stated preference methods allows the estimation of non-use or existence value and, consequently, estimates the total economic value of an environmental good or service.

Choice experiments (CE) involve introducing a set of hypothetical alternatives, each presenting a different situation with respect to some environmental amenity and other characteristics. Respondents are asked to rank the alternatives in order of preference or to pick the most preferred alternative. The rankings or choices can be analysed to determine the marginal rate of substitution between any characteristic and the level of the environmental amenity. If one of the characteristics is a monetary price then it is possible to compute the respondent's willingness to pay for the good or service of interest (Freeman, 2003). Because choice experiments are based on attributes, they allow the researcher to value attributes as well as situational changes. In the case of damage to a particular attribute, compensating amounts of other goods (rather than compensation based on money) can be calculated. An attribute-based approach is necessary to measure the type or amount of other 'goods' that are required for compensation (Navrud, 2004). This approach can provide more information about a range of possible alternative policies as well as reduce the sample size needed compared to contingent valuation. However, survey design issues with the CE approach are often more complex due to the number of goods that must be described and the statistical methods that must be employed (Navrud, 2004).

3.3.3 Evaluation of environmental impacts based on preferences revealed in political negotiations

The impact pathway requires an estimation of the impacts in physical terms and then a valuation of these impacts based on the preferences of the individuals affected. This approach has been successfully applied to human health impacts, for example, but in other areas this approach cannot be fully applied because data on valuation is missing (acidification and eutrophication of ecosystems) or because estimation of all physical impacts is limited (global warming).

Therefore and for these cases, a second best approach may be better than having no data or partial data. Therefore the use of approaches that elicit implicit values in policy decisions to monetise the impacts of acidification and eutrophication and of global warming has been explored. Marginal abatement costs would be equal to marginal damage costs if the emission limits imposed by environmental regulations were optimal. But policy makers do not know where the social optimum is, to say nothing about the twists and turns of the political processes that lead to the choice of regulations in practice. In reality the policy makers need information on damage costs, as provided by programmes such as ExternE, in order to formulate the environmental regulations. Therefore using abatement costs as proxy for damage costs begs the question. A general overview of methods and how they relate is given in Table 3.2.

Nonetheless abatement costs can be a valuable source of information for impacts whose monetary valuation has not yet been satisfactory or even possible, in particular global warming. This approach, called standard price or abatement cost approach, has also been tried for eutrophication and acidification, but the results for the latter impacts have not yet been included in the damage costs of ExternE because of problems with the data linking emissions and affected areas. The abatement cost approach is appropriate to the extent that the choices of policy makers correctly reflect the underlying values of the population. For impacts such as eutrophication and acidification, policy makers may have a better understanding of the values than the general population because they have the means to become well informed about the nature of the impacts whereas the general population lacks the necessary knowledge to have a well-informed opinion. Even though the results of the standard price or abatement cost approach must not be used for cost-benefit analysis and for environmental regulations, they can be used for comparing the external costs of different fuel chains, thus providing guidance for energy policy.

Table 3.2 Overview of methods used in ExternE to quantify and value impacts.

	Air pollution			Global warming
	Public health	Agriculture, building materials	Ecosystems	
ExternE, "Classical" impact pathway approach				
Quantification of impacts	Yes	Yes	Yes, critical loads	Yes, partial
Valuation	Willingness to pay (WTP)	market prices		Yes, WTP & market prices
Extension: Valuation based on preferences revealed in				
Political negotiations			UN-ECE; NEC	Implementing Kyoto, EU
Public referenda				Swiss Referenda

Even though the results for eutrophication and acidification have not yet been applied, it may be instructive to sketch very briefly how they were obtained. A reference scenario for the emissions of NO_x, SO_2, NH_3 and VOC was defined as the expected emissions in 2010 under business as usual, taking into account the legislation in force in Europe as of 1998. It is compared with three alternative scenarios for reducing these emissions: the Gothenburg Protocol, the initial proposal for the National Emission Ceilings Directive of the EU and the final version in which this directive was accepted. The abatement costs for reaching each of these three scenarios from the reference are available from the RAINS model of IIASA. The corresponding damage costs due to the impacts on health, agricultural crops, building materials and ozone formation have been calculated by ExternE. The benefits from reduced eutrophication and acidification have been calculated in physical units, as hectares saved. To derive the monetary value of a hectare saved, one needs to know what weighting factors the policy makers attached to the respective impact categories. That was done by an examination of the reasons given by the texts of the proposed or realised directives, supplemented by a questionnaire for policy makers. Combining the weighting factors of the policy makers with the respective impact categories, one obtains the benefits implicitly assigned by the policy makers. Since all the benefits except eutrophication and acidification are in monetary units, the monetary value of the latter follows by setting the sum of all benefits equal to the abatement cost. Of course, the uncertainties are large.

With regard to CO_2, an assessment of the costs for achieving Kyoto targets in the EU can be interpreted as a proxy for the collective willingness-to-pay in the EU for early action against global warming. For assessing technologies and fuel cycles in the mid-long-term, the best estimate is between €5-20/tCO_{2eq}, with the higher range reflecting the costs if emissions are controlled within Europe. For ExternE a value of €19/tCO_{2eq} has been selected. This number is also well below the penalty set in the emission trading scheme (€40/tCO_{2eq} for the first 3 years), which can be seen as an upper limit for the damage cost. A recent review showed that a value of €19/tCO_{2eq} is in the middle of the wider range of estimates, both from studies and from starting or experimental CO_2-trading schemes (Downing and Watkiss, 2003).

For another data point for CO_2 the results of referenda on energy taxes in Switzerland held in year 2000 have been analysed. Under plausible assumptions about the underlying WTP distribution, the average willingness of the Swiss population to pay energy taxes per kWh can be estimated. The referenda originally refer to taxes on non-renewable energy consumption in order to favour renewable energy. The change from fossil fuels to renewable energy affects mainly direct CO_2 emissions but not necessarily other pollutant emissions (e.g. NO_x or PM_{10} emission factors for biomass are comparable to those for fossil fuels). Therefore it is plausible to account the WTP per kWh fully to CO_2 as far as emissions are concerned. The resulting estimates are about €6 to 9/tCO_{2eq} for the geometric mean and about €14 to 22tCO_{2eq} for the arithmetic mean, confirming the plausibility of the value chosen by ExternE.

3.4 Benefit Transfer

According to Rosenberger and Loomis (2001), benefit transfer is defined as the adaptation and use of existing economic information derived for specific sites under certain resource and policy conditions to new contexts or sites with similar resources and conditions. Brouwer (1998) defines it as a technique where the results of monetary (environmental or health) valuation studies, estimated through market based or non-market based economic valuation techniques, are applied to a new policy context. Some authors (e.g. Navrud, 2004) prefer the term ‘value transfer’, since in many cases damage estimates can also be transferred from previous studies (termed study-sites) to new evaluation contexts (policy-sites).

Decision-makers often need economic analyses to support decisions among different policy alternatives. When the relevant economic values and the required resources are not available for developing new environmental valuation studies, then economic measures estimated in similar contexts and sites can provide a proxy for the estimates necessary for decision-making. In other words, benefit transfer is an alternative to fill in gaps in the availability of information on the preferences of individuals in a country or region. "Applying previous research findings to similar decision situations is a very attractive alternative to expensive and time consuming original research to quickly inform decision makers" (Brouwer, 1998).

3.4.1 Alternative benefit transfer methods

Navrud (2004) defines a typology of the most usual benefit transfer methods, identifying two main approaches. The unit value transfer approach, which involves the methods known as simple unit transfer (also known as single-point estimate or average-value transfer – Rosenberger and Loomis, 2001) and unit transfer with income adjustment, and the function transfer approach that uses the benefit function transfer method and meta-analysis (or meta-regression analysis).

Unit value transfer – simple unit transfer

This is the simplest method of transferring economic estimates from one site or context to another, based on using an estimate from a single relevant study-site or a range of point estimates if more than one study is considered relevant (average-value transfer). According to Navrud (2004), it assumes that the well-being experienced by an average individual at the original study-site will be equivalent to the well-being experienced by the average individual in the policy-site. Once this assumption holds, analysts can directly transfer the economic benefit or damage from the study-site to the policy-site. An alternative procedure, average-value transfer, is based on using a measure of the central tendency of relevant studies as the transfer estimate for a given policy-site. Rosenberger and Loomis (2001) argue that average value estimates, however, are no better than the data they are based on, that is, all of the eventual problems related to the credibility of any single estimate are also relevant for an average value based on that estimate. The authors claim that the primary steps to perform a single point estimate transfer (simple unit transfer) include identifying and quantifying the policy-induced changes, and locating and transferring a unit value (single estimate or average) representing the individuals' welfare measure.

An immediate limitation of this method is that individuals in the policy-site may differ from individuals at the study-site(s) in terms of socio-economic characteristics – income, education, religion, for example – that can affect their preferences. Therefore, Navrud (2004) concludes that the simple unit transfer approach should not be used for benefit transfer between countries with different income levels and costs of living.

Unit value transfer – unit transfer with income adjustment
The unit transfer with income adjustment method has been the most used practice for policy analysis in developing countries since most of the environmental valuation studies were conducted in developed countries (Navrud, 2004). This method assumes that the benefit value in the policy-site can be estimated by adjusting the benefit value in the study-site(s) by the ratio between income levels in both sites and the income elasticity of demand for the environmental good. Formally:

$$B_p = B_s.\left(\frac{Y_p}{Y_s}\right)^{\beta} \tag{3.1}$$

Where (B_p) is the adjusted policy-site benefit; (B_s) is the original benefit estimate in the study-site; (Y_p) and (Y_s) are the income levels; and (β) is the income elasticity of demand for the analysed environmental good.

However, it is argued that most studies assume GDP per capita as proxies for income in international benefit transfers, and income elasticity of demand equal to one. These common assumptions do not necessarily hold. Navrud (2004) argues that it is appropriate to use PPP estimates of per capita GDP, instead of GDP per capita, since these estimates are adjusted to reflect a comparable amount of goods and services that could be purchased with the per capita GDP in other country. Also, the author claims that there is no evidence that welfare measures associated with environmental goods vary proportionally with income, and sensitivity analyses should assume different levels of income elasticity of demand. Using an income elasticity equal to one would change the willingness-to-pay measure in the policy-site proportionally to the relative per capita income differential across the two areas of study, whilst income elasticity equal to zero would mean that no adjustment is considered for income differentials (Davis *et al.*, 1999).

Function transfer – benefit function transfer
Benefit-function transfer involves the use of a willingness-to-pay function, derived in a study-site preferably using stated or revealed preference techniques, which relates willingness to pay to a set of characteristics of the study-site population and the environmental good. That is, benefit function transfers use a model that statistically relates benefit measures with study factors such as characteristics of the user population and the resource being evaluated. The transfer process involves adapting the benefit function to the characteristics and conditions of the policy-site, forecasting a benefit measure based on this adaptation of the function, and use of the forecast measure for policy analysis (Rosenberg and Loomis, 2001).

The advantage of benefit function transfer, in contrast with unit value transfer, is that more information can be taken into account in the transfer process. When transferring a unit value estimate from a study-site to a policy-site, it is assumed that the two sites are identical across the various factors that determine the level of benefits derived in both sites. However, Rosenberger and Loomis (2001) argue that this is not always the case, their argument being based on different validity and reliability assessments of unit value transfers. The invariance involving the transfer of benefit measures alone makes these transfers insensitive or less robust to significant differences between the study-site and the policy-site. Therefore, the main advantage of transferring an entire benefit function to a policy-site is the apparently increased precision of tailoring a benefit measure to fit the characteristics of the policy-site.

Disadvantages of the method are primarily due to data collection and model specification in the original study. Navrud (2004) claims that the main problem with the benefit function approach relates to the exclusion of relevant variables in the willingness-to-pay function estimated in a single study. For example, when the estimation is based on observations from a single environmental good, the lack of variation in some of the independent variables avoids the inclusion of these variables in the model, and in another policy-site these variables may be important. Indeed, Rosenberg and Loomis (2001) report that factors in the benefit function may be relevant to the study-site but not to the policy-site. These factors can have distinct effects on the tailored benefit measures at a policy-site.

Function transfer – meta-analysis

Meta-analysis is used when the results of many valuation studies, developed in different study-sites, are used for estimating a single benefit transfer function. It is defined as the statistical summary of relationships between benefit estimates and quantifiable characteristics of studies. In meta-analysis, several studies are analysed as a group and each result of these studies is one observation in a regression analysis. The data for a meta-analysis are typically summary statistics from study-site reports and include quantified characteristics of the user population, the study site's environmental resources, and the valuation methodology used.

Navrud (2004) claims that meta-analysis allows analysts to evaluate the influence of a wider range of population and environmental good characteristics, as well as the modelling assumptions. The resulting regression equations can then be used to predict an adjusted unit value for the policy-site, given the availability of data on the independent variables for the policy-site. The meta-analysis regression has the welfare measure as dependent variable, the environmental good and population characteristics as independent variables (similar to the benefit function transfer), but also includes characteristics of the original studies in the study-sites. These characteristics include methodological variables, such as elicitation format, payment vehicle, and response rates in case of studies applying stated preference methods. However, the author argues that methodological variables are not particularly useful in predicting welfare estimates for environmental goods, especially in international benefit transfer, if we assume cross-country heterogeneity in preferences for environmental goods. The author concludes that, to increase the applicability of meta-analysis for benefit transfer, analysts should select original studies that are methodologically very similar to each other, isolating the effects of site and population characteristics on the estimates.

3.4.2 Validity and reliability

Several factors were identified that can affect the reliability and validity of benefit transfers. Rosenberger and Loomis (2001) summarised these factors:

- One group of factors that affects the validity of benefit transfers includes:
  - The quality of the original study greatly affects the quality of the benefit transfer process;
  - The limited number of studies investigating a specific environmental good, thus restricts the pool of estimates and studies from which to draw information;
  - The documentation of data collected and reported can be a limitation.
- A second group of factors is related to methodological issues. For example:
  - Different research methods may have been used across study-sites, including what question(s) was asked, how it was asked, what was affected by the management or policy action, how the environmental impacts were measured, and how these impacts affect recreation use;
  - Different statistical methods for estimating models can lead to large differences in values estimated. This also includes issues such as the overall impact of model misspecification and choice of functional form;
  - There are different types of values that may have been measured in primary research, including use values and/or passive- or non-use values.
- A third group of factors concerns the correspondence between the study site and the policy site, which arises because
  - Some of the existing studies may be based on valuing activities at unique sites and under unique conditions;
  - Characteristics of the study-site and the policy-site may be substantially different, leading to quite distinct values. This can include differences in quality changes, site quality, and site location.
- A fourth factor is the issue of temporality or stability of data over time. If the existing studies occurred at different points in time, relevant differences between then and now may not be identifiable nor measurable based on the available data.
- A fifth factor is the spatial dimension between the study-site and the policy-site. This includes the extent of the implied market, both for the extent and comparability of the affected populations and the resources impacted between the study-site and the policy-site.

These factors can lead to bias or error in the benefit transfer process, reducing its robustness. The objective of the benefit transfer process is to minimise mean square error between the true value and the predicted or transferred value of impacts at the policy-site. However, Rosenberger and Loomis (2001) claim that the original or true values are themselves approximations and are subject to error. Therefore, any information transferred from a study-site to a policy-site is accomplished with varying degrees of confidence in the applicability and precision of the information.

3.4.3 Validity tests

Studies have tested the validity and reliability of different benefit transfer methods and results have shown that the uncertainty in spatial and temporal benefit transfer can be large (e.g. Ready *et al.,* 2004; Kristoferson and Navrud, 2005). Although no standard protocol or guidelines for conducting benefit transfer is available, some studies compare benefit transfer estimates with contingent valuation studies of the same site to test the validity of benefit transfer. For example, Bergland *et al.* (1995), cited in Navrud (2004), conducted contingent valuation surveys for increased water quality in two different lakes in Norway, generated benefit functions for each of them, transferred the benefit function to the other, and then compared the transferred values with the original contingent valuation estimates. The authors also transferred and compared the mean (unit) values, since the lakes were rather similar in size and type of pollution problem. Several tests for transferability were conducted but transferred and original estimates were statistically different at the 5% level. However, the transfer error[5] varied between 20% and 40%, with predicted values being lower in one case (for one of the lakes) and higher for the other lake.

Ready *et al.* (2004) measured the benefits for specific health impacts related to air and water pollution in five European countries using similar contingent valuation surveys. The authors tested different benefit transfer methods against original contingent-valuation estimates, finding an average error of 38%. They concluded "accounting for measurable differences among countries in health status, income and other demographic measures, either through ad hoc adjustments to the transferred values or through value transfer function transfer, did not improve transfer performance" (Ready *et al.,* 2004). It suggests that cultural and attitudinal factors seem to be important in explaining differences in valuation across countries.

Navrud (2004) cites examples of validity tests performed within countries, across countries, and between developed and developing countries, and concluded that the results from these studies show that the uncertainty in value transfer can be large. The general indication is that benefit transfer cannot replace original studies, especially when the costs of being wrong are high.

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[5] Defined as the difference between transferred mean WTP and observed mean WTP, as a percentage of the observed mean WTP.

3.4.4 Conditions and limitations

Rosenberger and Loomis (2001) argue that some general conditions should be met to perform benefit transfers.

- The policy context should be carefully defined, identifying:
    - The extent, magnitude, and quantification of expected impacts from the proposed action;
    - The population that will be affected by the expected impacts;
    - The data needs, including the type of measure (unit, average, marginal value) and the degree of certainty surrounding the transferred data.
- The study-site data should also meet certain conditions:
    - Studies transferred must be based on adequate data, valid economic method, and correct empirical technique;
    - Contain information on the statistical relationship between benefits and socio-economic characteristics of the affected population;
    - Contain information on the statistical relationship between the benefits and physical/ environmental characteristics of the study site;
- The correspondence between the study-site and the policy-site should have the characteristics:
    - The environmental resource and the change in the quality or quantity of the resource at the study-site and the resource and expected change at the policy-site should be similar;
    - The markets for the study-site and the policy-site are similar, unless there is enough useable information provided by the study on own and substitute prices – other characteristics should be considered, including similarity of demographic profiles between the two populations and their cultural aspects;

3.5 Discounting

3.5.1 Discount rates

We do not need to rehearse again the rationale for discounting or the reasons for the continuing debate as to which rate(s) to use in the environmental context. Both are described in detail in European Commission (1995) and Friedrich and Bickel (2001). There are two ways in which a social discount rate can be derived. The first is the social rate of time preference (also known as the consumption discount rate), which attempts to measure the rate at which social welfare or utility of consumption falls over time. The social rate of time preference is given by:

$$i = z + n \times g \qquad (3.2)$$

where z is the rate of pure time preference (impatience – utility today is perceived as being better than utility tomorrow), g is the rate of growth of real consumption per capita, and n is the percentage fall in the additional utility derived from each percentage increase in consumption (n is referred to as the 'elasticity of the marginal utility of consumption').

The second reason is that, since capital is productive, a unit of a currency's worth of resources now will generate more than one unit of currency's worth of goods and services in the future. Hence an entrepreneur is willing to pay more than one unit in the future to acquire one unit's worth of these resources now. This argument for discounting is referred to as the 'private marginal opportunity cost of capital' argument, and for our purposes can be converted, in theory, to the social marginal opportunity cost of capital by subtracting external costs of the productive capital and adding the external benefits. In practice we often assume, for simplicity, that the two are the same.

In the presence of efficient markets and no taxes, the two measures would be equated by the market rate of interest. In practice the range of individual time preference rates is large and does not coincide with the rates for the opportunity cost of capital. As described in the source mentioned above, our solution to this is to suggest a range of values for the discount rates to be used. Depending on the assumptions made about the components of the social rate of time preference, values can reasonably be suggested in a range of 0% - 4.5%. These are set out in the table below

Table 3.3 Social time preference rates.

Pure Rate of Time Preference z	Elasticity of Marginal Utility of Consumption n	Per Capita Income Growth Rate g	Discount Rate $i = z + n \cdot g$
0	0	1.5	0
0	1	1.5	1.5
1.5	1	1.5	3
3	1	1.5	4.5

Values for the social opportunity cost of capital in the EU are generally found to average about 6%. Combining estimates for the social time preference rates with the social opportunity cost give a range of recommended discount rates for use in the ExternE project of:

- Low: 0%
- Central: 3%
- High: 6%

3.5.2 Theoretical rationale for declining discount rates

Weitzman (1998) points out that, when applying standard discounting methods to long-term effects, for any reasonable discount rates (above 1 or 2% per annum) what happens a few centuries from now hardly counts at all. There is therefore an issue as to what will be the deep-future real interest rates. For any period, the real rate of interest is determined by the productivity of investment, (the social marginal opportunity cost of capital referred to above), and for the deep future it is the same. By applying constant discount rates, economists are implicitly assuming that the productivity of investment will be the same in the deep future as in the recent past. Weitzman does not see fundamental reasons why this should not be so. But, the deep future is totally uncertain, and one of the most uncertain aspects of it is the discount rate itself. It is not the discount rate that should be probability-averaged over states of the world, but the discount factor. This makes a huge difference in the deep future, for very large time periods. Uncertainty about future interest rates provides a strong generic rationale for using certainty-equivalent social interest rates that decline over time from around today's market values down to the smallest imaginable rates for the far-distant future. This effect does not begin to operate until beyond the range of near future, in which we can be fairly confident today's rates will prevail.

His argument, then, is that when there is an uncertain discount rate, the correct discount rate for a particular time period – the certainty-equivalent discount rate – can be found by taking the average of the discount factor, rather than the discount rate itself. Table 3.4 illustrates this. Here, there are ten discount rate scenarios, with each scenario having an equal probability.

This shows that – in the limit – as the time period considered becomes larger and approaches infinity, the certainty-equivalent discount rate approximates the lowest discount rate being considered – in this case 1%. The empirical values given here are derived from a study by Newell and Pizer (2001), based on uncertainty in relation to US market interest rates on long-term government bonds using Weitzman's approach.

This profile of a declining discount rate over future time periods is not uncontroversial. There is, for example, no reason why we need to assume a fall in productivity growth. There is also no discussion of the social time preference rate. These issues are ripe for future research efforts. For the time being, we suggest that the range of constant rates outlined above be used in the first instance, and to use the Weitzman justification for a declining rate regime. Rounded values of those above would suggest the following: for about the next 25 years from the present, use a "low-normal" real annual interest rate of around 3-4%. For the period from about 25 to about 75 years from the present, use a within-period instantaneous interest rate of around 2%. For the period from about 75 to about 300 years from the present, use a within-period instantaneous interest rate of around 1%. And for more than about 300 years from the present, use a within-period instantaneous interest rate of around 0%.

Table 3.4 Uncertain discount factors and declining discount rates.

	Discount factors in year t				
Discount rate	10	50	100	200	500
1%	0.91	0.61	0.37	0.14	0.01
2%	0.82	0.37	0.14	0.02	0.00
3%	0.74	0.23	0.05	0.00	0.00
4%	0.68	0.14	0.02	0.00	0.00
5%	0.61	0.09	0.01	0.00	0.00
6%	0.56	0.05	0.00	0.00	0.00
7%	0.51	0.03	0.00	0.00	0.00
8%	0.46	0.02	0.00	0.00	0.00
9%	0.42	0.01	0.00	0.00	0.00
10%	0.39	0.01	0.00	0.00	0.00
Certainty-equivalent discount factor	0.61	0.16	0.06	0.02	0.00
Certainty-equivalent discount rate Newell and Pizer (2001)	4.73%	2.54%	1.61%	1.16%	1.01%

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4 Assessment of Impacts Caused by Emissions to Air, Water and Soil: The Impact Pathway Approach

4.1 Introduction

In order to calculate the damage costs (= external costs) of polluting activities such as energy production, one needs to carry out an impact pathway analysis (IPA), tracing the passage of a pollutant from where it is emitted to the affected receptors (population, crops, forests, buildings, etc.). As illustrated in Figure 4.1, the principal steps of an IPA can be grouped as follows:

- Emission: specification of the relevant technologies and pollutants, e.g. kg of oxides of nitrogen (NO_x) per GWh emitted by a power plant at a specific site);
- Dispersion: calculation of increased pollutant concentrations in all affected regions, e.g. incremental concentration of ozone, using models of atmospheric dispersion and chemistry for ozone formation due to NO_x (this step is also called environmental fate analysis, especially when it involves more complex pathways that pass through the food chain);
- Impact: calculation of the dose from the increased concentration, followed by calculation of impacts (damage in physical units) from this dose, using a dose-response function, e.g. cases of asthma due to this increase in ozone;
- Cost: economic valuation of these impacts, e.g. multiplication by the cost of a case of asthma.

The impacts and costs are summed over all receptors of concern. The work involves a multidisciplinary system analysis, with inputs from engineers, dispersion modellers, epidemiologists, ecologists and economists.

For many environmental choices one needs to look not only at a particular source of pollutants, but has to take into account an entire process chain by means of a life cycle assessment (LCA). For example, a comparison of power generation technologies involves an analysis of the fuel chain sketched in Figure 4.2. Whether an IPA of a single source or an LCA of an entire cycle is required, depends on the policy decision in question. For finding the optimal limit for the emission of NO_x from an incinerator, an IPA is sufficient, but the choice between incineration and landfill of waste involves an LCA.

In principle the damages and costs for each pollution source in the life cycle should be evaluated by a site-specific IPA. But in practice almost all LCA has taken the shortcut of first summing the emissions over all stages and then multiplying the result by site-independent impact indices. Also, most practitioners of LCA reject the concept of monetary valuation, preferring instead to use about ten non-monetary indicators of "potential impact" that are based on expert judgment.

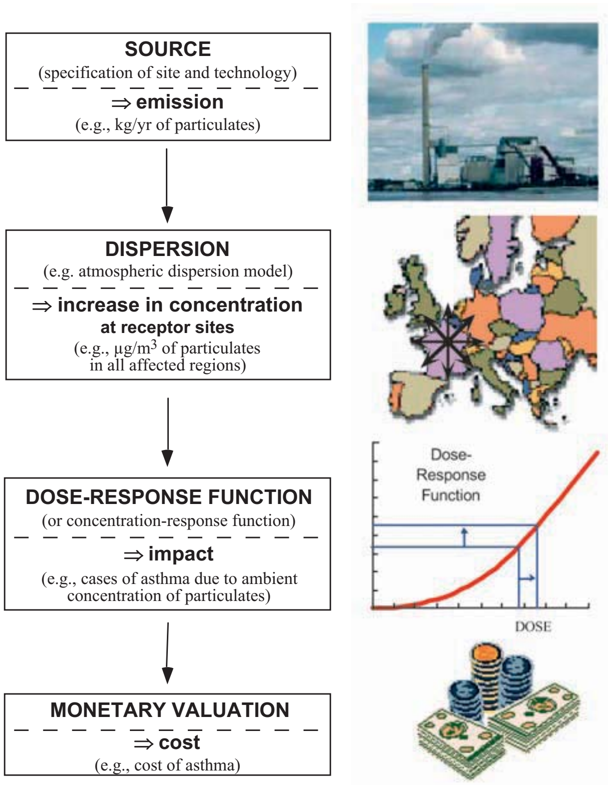

Figure 4.1 The principal steps of an impact pathway analysis, for the example of air pollution.

ExternE, however, has been using LCA in combination with IPA (impact pathway analysis) to get a complete assessment of external costs due to electricity production, including impacts that occur upstream and downstream of the power plant itself. That practice requires a modification if the external costs upstream or downstream have already been completely internalised. Of course, that is not the case at the present time for most pollutants and in most countries (SO_2 in Sweden being a good counter example).

The need to include upstream or downstream impacts in the external cost calculations arises from the lack of complete internalisation by the current environmental policies. If an external cost that arises upstream or downstream has already been internalised by an optimal pollution tax (i.e. a tax equal to the marginal damage) or by tradeable permits that are auctioned by the government, it should no longer be included – otherwise there would be double counting when the results are used, for example in a cost-benefit analysis or to determine the pollution tax for the power plant. On the other hand, for external costs that have been internalised by tradeable permits that are free, the residual damage has not been paid by the polluters and should be included in the analysis.

→ real impacts for each stage (site specific)

Goal: evaluate the entire matrix

Steps of impact pathway analysis → **Stage of fuel chain** ↓	Emission	Dispersion	Exposure-response function	Economic valuation
Fuel extraction				
Fuel transport				
Power plant				
Transmission of electricity				
Management of wastes				

Life cycle assessment: first sum over emissions ↓ Σ then → × multiplication by **"potential impact"** indices

Figure 4.2 Relation between impact pathway analysis and current practice of most LCA, illustrated for the example of electricity production. From Spadaro and Rabl (1999).

And, of course, the contributions upstream or downstream should be indicated separately, to avoid misuse when the results are used for regulations that concern a power plant. For example, it would not make sense to tax a power plant for damage caused by a coal mine in a different country (if all polluters had to pay a tax corresponding to the full LCA impacts, there would be double taxation).

The reader may wonder about the relation between an IPA and an environmental impact study (EIS) that is required before a proposed installation (power plant, incinerator, factory, etc.) can be approved. The purpose of an EIS is to ensure that nobody is exposed to an unacceptable risk or burden. Since the highest exposures are imposed in the local zone, it is sufficient for an EIS to focus on a local analysis, up to perhaps ten km depending on the case. Thus an EIS provides the possibility of a veto if a proposed installation is considered unacceptable. In contrast the calculation of total damage costs requires an IPA where the damages are summed over all affected receptors (for most air pollutants emitted in Europe that is the entire continent, and for greenhouse gases it is the entire globe). Damage costs are needed primarily by decision makers at the national or international level, or generally by anyone concerned with total impacts.

4.2 Dispersion of Pollutants and Exposure

The principal greenhouse gases, CO_2, CH_4 and N_2O, stay in the atmosphere long enough to mix uniformly over the entire globe. No specific dispersion calculation is needed but the calculation of impacts is extraordinarily complex, see the documentation published by the Intergovernmental Panel on Climate Change (IPCC http://www.ipcc.ch). For most other air pollutants, in particular PM_{10} (particulate matter with diameter less than 10 μm), NO_x and SO_2, atmospheric dispersion is significant over hundreds to thousands of km, so both local and regional effects are important. ExternE uses therefore a combination of local and regional dispersion models to account for all significant damages. The main models for the local range (< 50 km from the source) have been the gaussian plume models ISC (Brode and Wang, 1992) for point sources such as power plants, and ROADPOL for lines sources (emissions from transport) (Vossiniotis *et al.,* 1996).

At the regional scale one needs to take into account the chemical reactions that lead to the transformation of primary pollutants (i.e. the pollutants as they are emitted) to secondary pollutants, for example the creation of sulphates from SO_2. Here ExternE uses the Windrose Trajectory Model (WTM) (Trukenmüller and Friedrich, 1995) to estimate the concentration and deposition of acid species. WTM is a user-configurable Lagrangian trajectory model, derived from the Harwell Trajectory model (Derwent and Nodop, 1986). The modelling of ozone is based on the EMEP MSC-W oxidant model (Simpson *et al*., 1992; Simpson and Eliassen, 1997). EMEP is the official model used for policy decisions about transboundary air pollution in Europe.

Several tests have been carried out to confirm the accuracy of the results. For example, we have checked the consistency between ISC and ROADPOL, and we have compared the concentrations predicted by WTM with measured data and with calculations of the EMEP programme.

Whereas only the inhalation dose matters for the classical air pollutants (PM_{10}, NO_x, SO_2 and O_3), toxic metals and persistent organic pollutants also affect us through food and drink. For these a much more complex IPA is required to calculate ingestion doses. During the NewExt phase of ExternE (see ExternE, 2004) two models were developed for the assessment of external costs due to the emission of the most toxic metals (As, Cd, Cr, Hg, Ni and Pb), as well as certain organic pollutants, in particular dioxins. They take into account the pathways in Figure 4.3. One of these models ("WATSON") is a multi-zonal model that links the regional air quality model of EcoSense to a soil and water multimedia model of the Mackay level III/IV type. The other model (Spadaro and Rabl, 2004) is based mostly on transfer factors published by EPA (1998), with some supplemental data of IAEA (1994 and 2001). These transfer factors account in a simple manner for the transport of a pollutant between different environmental compartments, for example the uptake by agricultural crops of a pollutant from the soil. The uncertainties of these models are large, but at least one has approximate values for the pollutants of concern here. The results published by ExternE are based on both of these models.

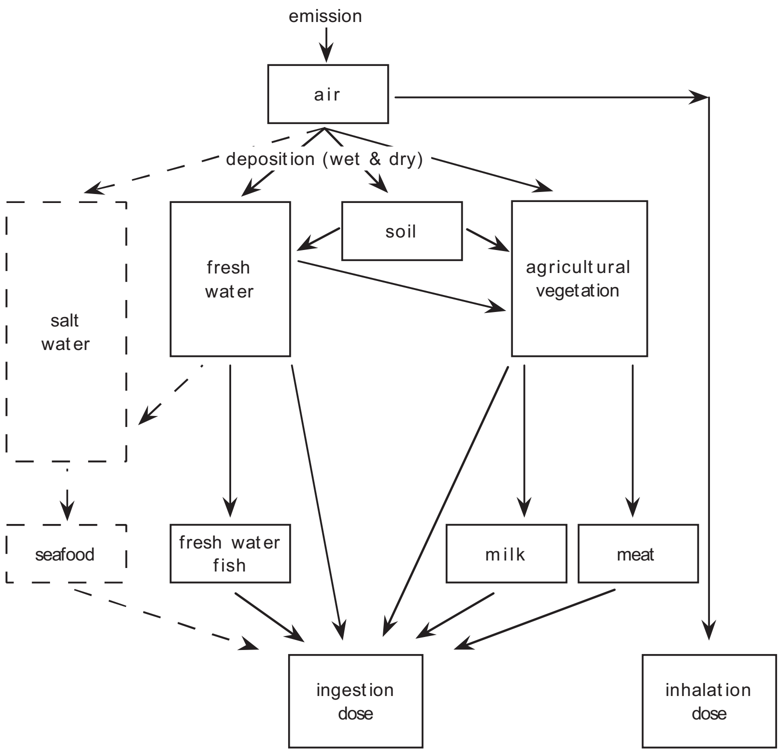

Figure 4.3 Pathways taken into account for health impacts of air pollutants. Direct emissions to soil or water are a special case where the analysis begins at the respective "soil" and "water" boxes. The impacts from seafood have not yet been calculated.

We do not yet have all the elements for calculating the dose due to ingestion of seafood, which is potentially large because of bioconcentration and because most fish comes from the ocean rather than freshwater. Even if the concentration increment in the sea is very small, the collective dose from seafood could be significant if the removal processes (sedimentation) are slow and the analysis has no cut-off in time.

A general result of this analysis is that, when these pollutants are emitted into the air, the ingestion dose can be about two orders of magnitude larger than the dose by inhalation. Because nowadays most food is transported over very large distances, the total dose does not vary much with the site where these pollutants are emitted into the air. As far as damages are concerned, one has to note that the same dose can have a very different effect on the body depending on whether it is inhaled or ingested. Cd, Cr-VI and Ni, for instance, are according to current knowledge carcinogenic only through inhalation.

4.3 Dose-Response Functions

4.3.1 General considerations

The dose-response function (DRF) relates the quantity of a pollutant that affects a receptor (e.g. population) to the physical impact on this receptor (e.g. incremental number of hospitalisations). In the narrow sense of the term, it should be based on the dose actually absorbed by a receptor. However, the term dose-response function is often used in a wider sense where it is formulated directly in terms of the concentration of a pollutant in the ambient air, accounting implicitly for the absorption of the pollutant from the air into the body. The functions for air pollutants are typically of the that kind, and the terms exposure-response function or concentration-response function (CRF) are often used.

The DRF is a central ingredient in the impact pathway analysis and merits special attention. A damage can be quantified only if the corresponding DRF is known. Such functions are available for the impacts on human health, building materials, and crops, caused by a range of pollutants such as primary and secondary particles (i.e. nitrates, sulphates), ozone, CO, SO_2, NO_x, benzene, dioxins, As, Cd, Cr, Ni and Pb. The most comprehensive reference for health impacts is the IRIS database of EPA (http://www.epa.gov/iriswebp/iris/index.html). For the application in an IPA, that information often has to be expressed in somewhat different form, accounting for additional factors such as the incidence rate. Unfortunately, for many pollutants and many impacts the DRFs are very uncertain or not even known at all. For most substances and non-cancer impacts the only available information covers thresholds, typically the NOAEL (no observed adverse effect level) or LOAEL (lowest observed adverse effect level). Knowledge of thresholds is not sufficient for quantifying impacts; it only provides an answer to the question whether or not there is a risk. The principal exceptions are carcinogens and the classical air pollutants, for which explicit DRFs are known (often on the assumption of linearity and no threshold).

By definition a DRF starts at the origin, and in most cases it increases monotonically with dose, as sketched schematically in Figure 4.4. At very high doses the function may level off in S-shaped fashion due to saturation, but that case is not of interest here. DRFs 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.

A major difficulty for health impacts lies in the fact that one needs relatively high doses in order to obtain observable non-zero responses unless the sample is very large; such doses are usually far in excess of typical ambient concentrations in the EU or North America. Thus there is a serious problem of how to extrapolate from the observed data towards low doses. Figure 4.4 indicates several possibilities for the case where the point P corresponds to the lowest dose at which a response has been measured. The simplest is the linear model, i.e. a straight line from the origin through the observed data point(s). The available evidence suggests that a dose-response function is unlikely to go above this straight line in the low dose limit. But the straight line model does appear to be appropriate in many cases, in particular for many cancers. In fact, most estimates of cancers due to chemicals or radiation assume this linear behaviour.
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.

There is even the possibility of a "fertiliser effect" at low doses, as indicated by the dashed line in Figure 4.4. This can be observed, for example, in the dose-response functions for the impact of NO_x and SO_2 on crops: a low dose of these pollutants can increase the crop yield, in other words the damage is negative. Generally a fertiliser effect can occur with pollutants that provide trace elements needed by an organism.

In practice most DRFs used by ExternE, in particular all the ones for health, are assumed to be linear (without threshold). Note that 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; only the slope matters. For particles, NO_x, SO_2, O_3 and CO the background in most countries is above the level where effects are known to occur. Thus the precise form of the ER function at extremely low doses is irrelevant for these pollutants; if there is a no-effects threshold, it is below the background concentrations of interest.

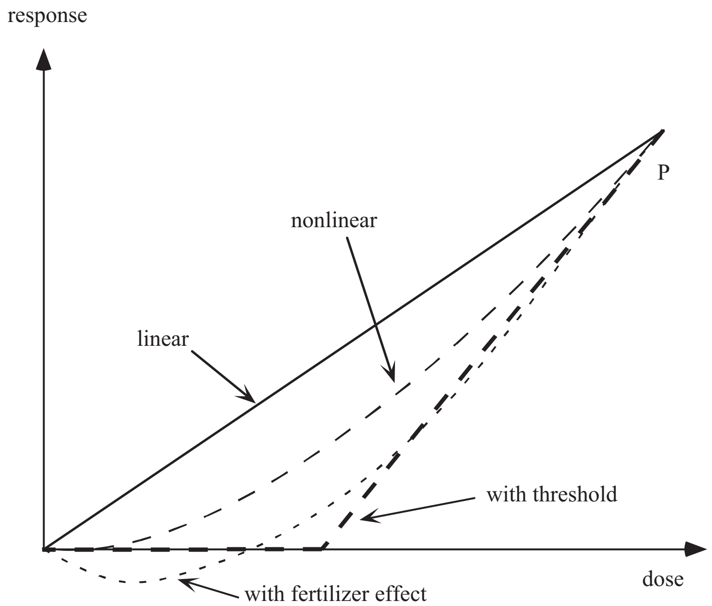

Figure 4.4 Possible behaviour of dose-response functions at low doses. If P is the lowest dose where a non-zero impact has been observed, the extrapolation to lower doses is uncertain but values higher than linear are unlikely.

4.3.2 Health Impacts

In terms of costs, health impacts contribute the largest part of the damage estimates of ExternE. A consensus has been emerging among public health experts that air pollution, even at current ambient levels, aggravates morbidity (especially respiratory and cardiovascular diseases) and leads to premature mortality (e.g. Wilson and Spengler, 1996, or the AIRNET website http://airnet.iras.uu.nl). There is less certainty about specific causes, but most recent studies have identified fine particles as a prime culprit; ozone has also been implicated directly. The most important cost comes from chronic mortality due to particles, calculated on the basis of Pope *et al.* (2002) (this term, chosen by analogy with acute and chronic morbidity impacts, indicates that the total or long-term effects of pollution on mortality have been included, in contrast to acute mortality impacts, which are observed within a few days of exposure to pollution). Another important contribution comes from chronic bronchitis due to particles (Abbey *et al.,* 1995). In addition there may be significant direct health impacts of SO_2, but for direct impacts of NO_x the evidence is less convincing.

In ExternE the working hypothesis has been to use the DRFs for particles and for O_3 as the basis. The health impacts of NO_x and SO_2 are assumed to arise indirectly from the particulate nature of nitrate and sulphate aerosols, and they are calculated by applying the particle DRFs to these aerosol concentrations. But the uncertainties are large because there is insufficient evidence for the effects of the individual components or characteristics (acidity, solubility, …) of particulate air pollution. In particular there is a lack of epidemiological studies of nitrate aerosols because until recently this pollutant has not been monitored by air pollution monitoring stations. All DRFs for health impacts have been assumed to be linear at the population level, in view of the lack of evidence for thresholds at current ambient concentrations. In contrast to the homogeneous populations of cloned animals studied by toxicologists, the absence of a no-effect threshold is plausible for real populations because they always contain individuals with widely differing sensitivities (for example, at any moment about 1% is within the last nine months of life and thus extremely frail).

4.4 Monetary Valuation

4.4.1 General considerations

The goal of the monetary valuation of damages is to account for all costs, market and non-market. For example, the valuation of an asthma attack should include not only the cost of the medical treatment but also the willingness-to-pay (WTP) to avoid the residual suffering. It turns out that damage costs of air pollution are dominated by non-market goods, especially mortality. If the WTP for a non-market good has been determined correctly, it is like a price, consistent with prices paid for market goods. Economists have developed several tools for determining non-market costs. Of these tools contingent valuation (CV) has enjoyed increasing popularity in recent years (Mitchell and Carson, 1989). The results of well conducted studies are considered sufficiently reliable. However, CV studies are not the only instruments that can be used for deriving monetary values. There are other valuation methods that can be used in addition or complementarily.

4.4.2 Mortality

The cost of mortality is usually evaluated by means of the value of a prevented fatality (VPF), often called "value of statistical life" (VSL), an unfortunate term that often evokes hostile reactions among non-economists. In reality VPF is merely a shorthand for "willingness-to-pay (WTP) to avoid the risk of an anonymous premature death". WTP (including ability to pay) is limited, even if we feel that the value of life is infinite – to save an individual in danger, no means are spared. Typical values recommended for policy decisions in Europe and North America are in the range of €1 to 5 million. Previous phases of ExternE (see European Commission, 1999a-d; ExternE, 2000) had used values around €3 million, chosen as average of the VPF studies that had been carried out in Europe. More recently ExternE (2004) carried out a new CV study and lowered the value to €1 million.

But whereas VPF is relevant for accidental deaths, it is not appropriate for air pollution mortality; the latter is primarily cardio-pulmonary and the associated loss of life expectancy (LE) per premature death is much shorter than for accidents. Furthermore, one can show (Rabl, 2003) that the total number of premature deaths due to air pollution cannot even be determined. One of the reasons is that air pollution cannot be identified as cause of any individual death; it is only a contributory, not a primary cause of death. Epidemiological studies of total (as opposed to acute) air pollution mortality cannot distinguish whether the observed result is due to a few people suffering a large loss of LE or many suffering a small loss. It is quite plausible that everybody's life is shortened to some extent by pollution, in which case every death would be a premature death due to pollution. Number of deaths is therefore not a meaningful indicator of the total air pollution mortality (even though several authors who do not understand this point have published numbers). Rather one has to use loss of LE which is indeed a meaningful indicator.

For the valuation of LE loss one needs the value of a life year (VOLY). In contrast to hundreds of VPF studies that have been carried out in many industrialised countries, VOLY has received little attention until recently. A significant step forward was taken by Krupnick *et al.* (2002) who developed a questionnaire specifically for the CV of air pollution mortality which they have applied in several countries (Canada, Japan and USA). More recently this questionnaire has also been applied in France, Italy and the UK (ExternE, 2004). The application in France (Desaigues *et al.,* 2004) involved not only the original questionnaire of Krupnick *et al.* but also the test of several variants, in particular variants that phrased the elicitation question directly in terms of LE gain (rather than risk of dying as in the original version), a formulation that is being used for the valuation work in the current phase of ExternE (the Integrated Project NEEDS). A crucial point that needs to be explained very carefully in such a questionnaire is that air pollution mortality does not cut off a few months of misery at the end of life but causes "accelerated ageing". Based on the results in France, Italy and the UK, ExtenE is now using a VOLY of €50,000.

4.5 Software

For the calculation of damage costs ExternE uses the EcoSense software package, an integrated impact assessment model that combines atmospheric models (WTM and ISC) with databases for receptors (population, land use, agricultural production, buildings and materials, etc.), dose-response functions and monetary values.

In addition there are two tools for simplified approximate assessments: EcoSenseLE and RiskPoll. EcoSenseLE (Look up Edition) provides tables of typical damage costs for a variety of emission sites. RiskPoll is a package of several models with different input requirements and levels of accuracy. It is based on the interpolation of dispersion calculations by EcoSense and, with its simplest version (the "uniform world model" described in section 11.2.2), yields results that are typically within a factor of two to three of detailed EcoSense calculations for stack heights above 50 m. A more complex model of RiskPoll includes the ISC gaussian plume model for the analysis of local impacts and emissions at or near ground level. RiskPoll also contains a module for the multimedia pathways of Figure 4.3.

Information on these software tools can be found at the ExternE website (http://www.externe.info). EcoSense can be obtained by paying a small handling fee after signing a license agreement. EcoSenseLE is an online tool at this website, and RiskPoll can be downloaded without charge or restrictions.

4.6 Calculation of Marginal Damage for Non-linear Impacts

The goal of ExternE is to estimate marginal damage costs because the socially optimal level of pollution control corresponds to the point where the sum of marginal damage cost and marginal abatement cost equals zero. However, if this seemingly simple statement is interpreted carelessly it could lead to absurd policy recommendations for impacts that are a non-linear function of the emission. To illustrate this problem, consider Figure 4.5 which shows a pollutant whose damage increases with emission at low emission levels but decreases again if the emission is high. Such a situation actually occurs with O_3 impacts as a function of one of the precursor emissions, NO (note that most NO_x is emitted as NO). The case of O_3 damage due to NO is the most extreme (complicated even more by the strong dependence of the curve on the other precursor VOC), but the problem also occurs in milder form with aerosols created by NO_x and SO_2 emissions.

With a careless interpretation one would find a negative marginal damage (tangent at the current emission level E1), implying that the policy response should be to encourage even greater emission of this pollutant. Such a policy response would miss the real optimum at E_{opt}. To provide the correct information to policy makers, one needs to examine carefully what the marginal damage costs will be used for and how they should be calculated. In fact, the correct calculation depends on the use of the results.

Probably the most important use of ExternE is the formulation of policies (e.g. pollution taxes or tradeable permits) to reduce the emissions to their social optimum. For this application the key observation is that the optimisation condition (marginal damage cost + marginal abatement cost = 0) requires knowledge of these marginal costs in the vicinity of the optimal emission level. Both the damage cost and the abatement cost can vary with emission site, and so does the optimal emission level. Ideally a policy maker should know the entire cost curves for marginal damage and abatement at each site. In the case of NO_x, SO_2 and VOC the damage costs are complicated site-dependent functions of not only the pollutant under consideration but also the simultaneous emission of several other pollutants with due consideration of all of their respective emission sites. The optimisation requires the solution of the coupled optimisation equations.

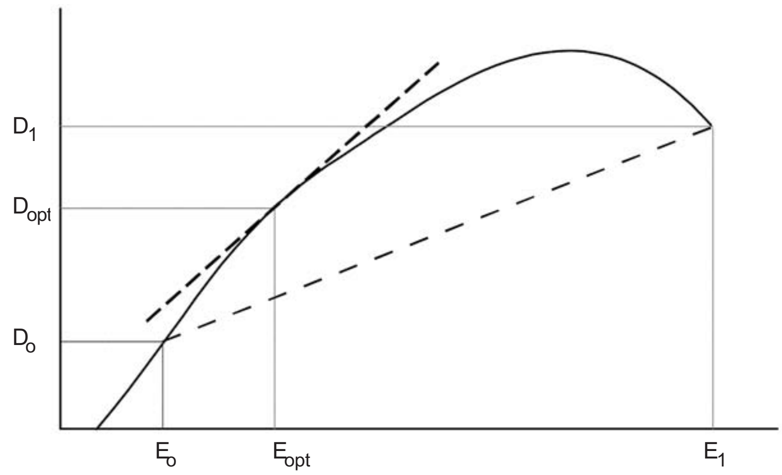

Figure 4.5 Pollutant whose damage D increases with emission E at low levels but decreases again if the emission is high. Slope of thick dashed line is the appropriate marginal damage, i.e. at optimal emission (unknown). Slope of chord from pre-industrial (E_0, D_0) to current (E_1, D_1) would be a better estimate of the appropriate marginal damage than the marginal damage at current emission E_1.

A first estimate of something like an ‚optimal emission level' has been estimated for the years 2010 and 2020 within the so-called CAFE (clean air for Europe) process, where efficient scenarios for pollution control in the EU have been created by integrated assessment models, especially by the RAINS model operated by IIASA and the model MERLIN from IER Stuttgart. These scenarios could be used as background scenarios.

The optimal NO_x emissions are much more uncertain than those for SO_2, for several reasons. Not only is the damage cost due to nitrate aerosols uncertain because of the lack of information on their toxicity, but the optimum depends also on the damage costs due to O_3, because the optimisation for NO_x involves setting the marginal abatement cost equal to the total marginal damage cost, not the individual cost components due to nitrates and ozone. The O_3 damage due to NO_x depends in turn on the background emissions of VOC. So far the optimal emission levels for VOC have not been estimated, and in any case iterations would be needed because of the coupled nature of the equations.

To conclude, the marginal damage costs of ExternE have to be calculated with emissions inventories that are much closer to the optimal emission levels than those that have been used until now. That will have a major effect on the results. Since the optimal emission levels are not known, the process is iterative. Fortunately there seems to be a fair amount of tolerance to errors in the determination of the optimal emissions, as shown by Rabl, Spadaro and van der Zwaan (2005), so even an initial estimation of the optimum may suffice for the purpose of calculating the damage costs of ExternE.

4.7 The Effect of Uncertainties and Ways to Address Uncertainties

Damage cost estimates are notorious for their large uncertainties (Rabl and Spadaro, 1999), and many people have questioned the usefulness of damage costs. The first reply to this critique is that even an uncertainty by a factor of three is better than infinite uncertainty. Second, in many cases the benefits are either so much larger or so much smaller than the costs that the implication for a decision is clear even in the face of uncertainty. Third, if policy decisions are made without a significant bias in favour of either costs or benefits, some of the resulting decisions will err on the side of costs, others on the side of benefits. Rabl, Spadaro and van der Zwaan (2005) have examined the consequences of such unbiased errors and found a very reassuring result: the extra social cost incurred because of uncertain damage costs (compared to the minimal social cost that one would incur with perfect knowledge) is remarkably small, less than 10 to 20% in most cases even if the damage costs are in error by a factor three. However, without any knowledge of the damage costs, the extra social cost could be very large.

One possibility to explore the uncertainties in the context of specific decisions is to carry out sensitivity analyses and to check whether the decision (e.g. implementation of technology A instead of technology B) changes for different assumptions (e.g. discount rate, costs per tonne of CO_2, valuation of life expectancy loss). It is remarkable that certain conclusions or choices are robust, i.e. do not change over the whole range of possible values of external costs. Furthermore, it can be shown that the ranking of electricity production technologies, for example, with respect to external costs does not change if assumptions are varied. A further option is to explore how much key values have to be modified before conclusions change. It can then be discussed whether the values triggering the change in decision can be considered realistic or probable.

A considerable share of uncertainties is not of a scientific nature (data and model uncertainty) but results from ethical choices (e.g. valuation of lost life years in different regions of the world) and uncertainty about the future. One approach to reduce the range of results arising from different assumptions on discount rates, valuation of mortality, etc. is to reach agreement on (ranges of) key values. Such “conventions for evaluating external costs”, resulting from discussion of the underlying issues with relevant social groups or policy makers, would help in narrowing the range of costs obtained in sensitivity analyses. This would help to make decision making in concrete situations easier and to focus on the remaining key issues to be solved in a specific situation.

4.8 Presentation of Results

The multitude of uncertainties described in the previous section makes the presentation of results a challenging task. ExternE does give estimates of the uncertainty, but sometimes they are not prominently placed together with the central estimate. Showing a result together with an explanation of its uncertainty is more difficult than showing a simple number. Finding the most appropriate way to communicate the uncertainties is not easy, especially since different users have different information needs.

Furthermore there are gaps in what currently can be quantified. Potentially important gaps should be reported together with the results. The problem is how to judge which impacts are potentially important, e.g. might have significant damage costs, and how to represent them. At the start of the analysis, ExternE used a screening process, analysing the ubiquity, irreversibility and persistency of a potential impact, and this screening process should continue. As with any assessment method, there may be other important impacts that have not yet been recognised as such (and ideal decision-making would take this eventuality into account).

Following recommendations can be given with respect to the presentation of results in order to ensure transparency:

- Present not only a single monetary value, but results for different subcategories (e.g. human health impacts, crop losses; or by pollutant).
- Present not only monetary values but as well physical impacts for important impact categories (e.g. number of life years lost).
- Carry out sensitivity analyses: present results for alternative assumptions for: VOLY, CO_2 damage/abatement cost, CRF for chronic motality (toxicity of primary and secondary particles).
- Describe gaps in the analysis.

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5 Impact Pathway Approach: Models for Pollutant Dispersion and Sound Propagation

5.1 Airborne Pollutants

This section draws on the relevant chapters of the 1998 update of the ExternE methodology report (European Commission, 1999), partly updated and extended by Bert Droste-Franke.

5.1.1 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 such a way that they are a *model-interpreter* rather than a *model*. Model specifications such as chemical equations, dose-response functions or monetary values, for example, 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.

5.1.2 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 metres 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 5.1). 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. The impacts of photochemical formation from primary emissions have to be considered also on a regional scale. For this analysis a parameterised Lagrangian Ozone model is used. Alternatively, Eulerian models can be applied directly or in a parameterised form. Options to use such models on the European level are currently being analysed and will be considered for future assessments. Due to the modular structure of EcoSense, it is possible to integrate new dispersion models as they become available.

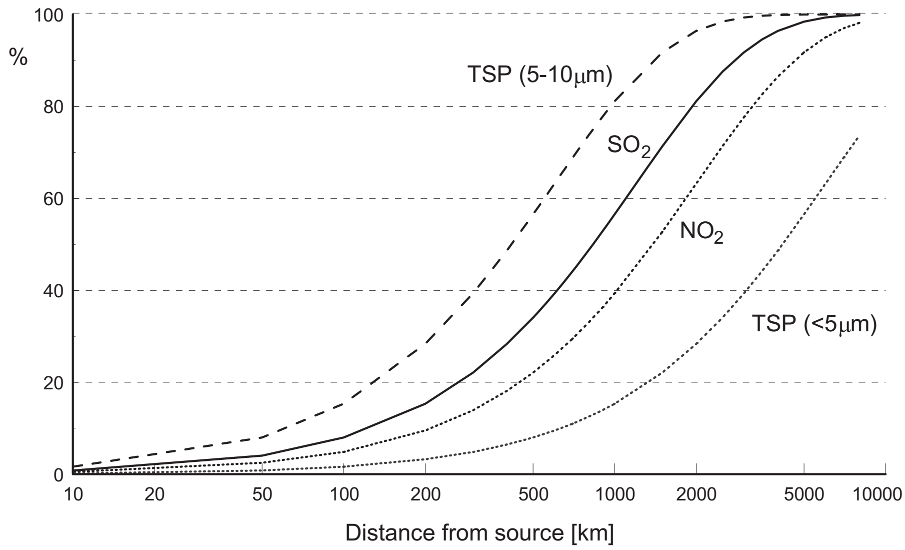

Figure 5.1 Percentage of cumulative damage expected with distance from the emission source.

5.1.3 Scope of the EcoSense model

EcoSense was developed to support the assessment of priority impacts resulting from the exposure to airborne pollutants, namely impacts on health, crops, building materials, forests and ecosystems. Although global warming is certainly among the priority impacts related to air pollution, this impact category is not covered by EcoSense because of the very different mechanism and global nature of impact. Priority impacts like occupational or public accidents are not included either because the quantification of such impacts is based on the evaluation of statistics rather than on modelling. Version 4.01 of EcoSense covers 14 pollutants, including the 'classical' pollutants SO_2, NO_x, particulates, CO and ozone, as well as some of the most important heavy metals and hydrocarbons, but does not include impacts from radioactive nuclides. The description in this chapter focuses on the most up-to-date Version 4.01 of EcoSense designed for the analysis of single energy sources in Europe. Further versions of EcoSense are operated at IER including versions designed for the analysis of road transport and multiple sources such as whole source sectors and countries in Europe and EcoSense versions transferred to regions outside Central Europe, namely Brazil/South America, China/Asia, Russia, and Ukraine.

5.1.4 The EcoSense modules

Figure 5.2 shows the modular structure of the EcoSense model. All data – input data, intermediate and final results – are stored in a relational database system. The two air quality models integrated in EcoSense are stand-alone linked to the system by pre- and postprocessors. There are individual executable programs for each of the impact pathways, which make use of common libraries.

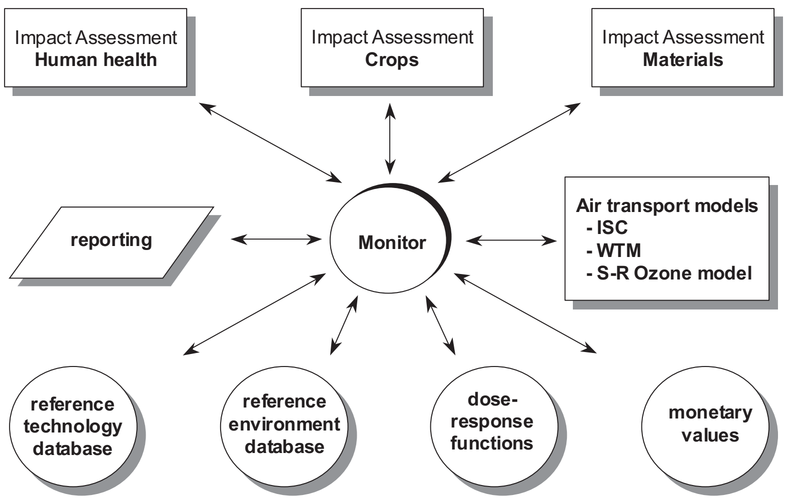

Figure 5.2 Structure of the EcoSense model

Calculations usually start with modifications of input data in the databases provided by the model, shown as circles in Figure 5.2. These hold technology and environmental data for the reference technology, receptor data (reference environment database), dose-response functions and monetary values applied for the model. In a second step, air quality modelling is carried out with the models specified by the user before, in a third step, the impact assessment modules are started in which selected exposure-response functions are used together with selected monetary values to estimate physical impacts and damage costs. Effects on human health, crops and materials are assessable. Finally, the calculated results can be compiled as a 'readable' report and a table text file. The latter is provided in a format which can easily be imported as a table into MS Excel. Furthermore, individual results can be displayed on a map. Two geographical structures are used for data processing: for the input of geographical data, administrative units down to municipality level for some regions and, for air quality modelling, (polar-stereographic) grids with maximum resolutions of 10 x 10 km^2 (local) and 50 x 50 km^2 (regional/European-wide).

5.1.5 The air quality models integrated in EcoSense

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. A frequently 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:

$$c(x,y,z) = \frac{Q}{u 2\pi \sigma_y \sigma_z} \cdot \exp\left[-\frac{y^2}{2\sigma_y{}^2}\right] \cdot \left(\exp\left[-\frac{(z-h)^2}{2\sigma_z{}^2}\right] + \exp\left[-\frac{(z+h)^2}{2\sigma_z{}^2}\right] \right) \tag{5.1}$$

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 in this type of model include those of idealised terrain and meteorological conditions so that the plume travels with the wind in a straight line. Dynamic features that affect the dispersion, for example vertical wind shear, are ignored. These assumptions generally restrict the range of validity of the application of these models to the region within some 50 km of the source. The straight line assumption is justified for a statistical evaluation of a long period, where mutual changes in wind direction cancel each other out, rather than for an evaluation of short episodes.

EcoSense employs the Industrial Source Complex Short-term model, version 2 (ISCST2) of the U.S. EPA (Brode and Wang, 1992). The model calculates hourly concentration values of SO_2, NO_x and particulate matter for one year at the centre of each small EUROGRID cell in a 10 x 10 km^2 grid centred on the site of the plant. Effects of chemical transformation and deposition are neglected. Annual mean values are obtained by temporal averaging of the hourly model results.

The σ_y and σ_z diffusion parameters are taken from BMJ (1983). This parameterisation is based on the results of tracer experiments at emission heights of up to 195 m (Nester and Thomas, 1979). More recent mesoscale dispersion experiments confirm the extrapolation of these parameters to distances of more than 10 km (Thomas and Vogt, 1990).

The ISCST2 model assumes reflection of the plume at the mixing height, i.e. the top of the atmospheric boundary layer. It also provides a simple procedure to account for terrain elevations above the elevation of the stack base:

- The plume axis is assumed to remain at effective plume stabilisation height above mean sea level as it passes over elevated or depressed terrain.
- The mixing height follows the terrain.
- The effective plume stabilisation height h_{stab} at receptor location (*x,y*) is given by:

$$h_{stab} = h + z_s - \min(z|_{(x,y)}, z_s + h_s) \tag{5.2}$$

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.

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, 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.

Several different types of model have been used to investigate acid deposition. These include Eulerian grid models, Lagrangian trajectory models and statistical models. These have been discussed in detail by several authors (Johnson, 1983; Eliassen, 1980, 1984; Hough and Eggleton, 1986; Schwartz, 1989). Lagrangian models, such as the Windrose Trajectory Model incorporated into EcoSense, consider air parcels that move with the direction and velocity of the wind. Eliassen (1984) provides a review of some aspects of Lagrangian models of air pollution. There are two main types of these models; those orientated towards the source of pollution and those that are receptor-orientated. In the first case, the source provides an initial mass of pollutant to the model air parcel, which subsequently moves away from the emission site. In the receptor-orientated case, the air parcel moves over various emission sources until it arrives at the receptor site. Lagrangian models permit the inclusion of more detailed chemistry than the Eulerian schemes, but the role of mixing between air parcels with different origins is not included. The effects of wind shear, which give different trajectory paths to parcels of air in different levels in the atmosphere, is seldom considered as the common assumption is that most of the pollution is confined to the mixing layer. Nevertheless, despite these theoretical problems, Lagrangian models have proved useful because their sensitivity to individual emission contributions can be rapidly assessed. Indeed, Lagrangian type models have proved capable of reproducing the distribution pattern and magnitude of regional sulphate deposition (Schwartz, 1989).

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 Lagrangian plume model employing an air parcel with a constant mixing height of 800 m moving with a representative wind speed. The results are obtained at each receptor point by considering the arrival of 24 trajectories weighted by the frequency of the wind in each 15° sector. The trajectory paths are assumed to be along straight lines and are started at 96 hours from the receptor point. The chemical scheme of the model is shown in Figure 5.3.

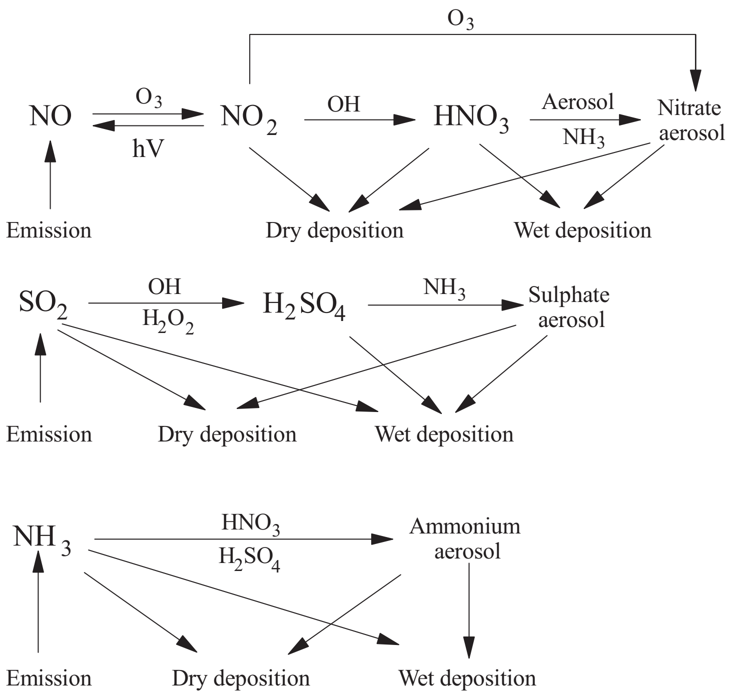

Figure 5.3 Chemical reactions of the sulphur and nitrogen species included in the Harwell Trajectory Model.

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 Synthesising Centre-West of EMEP at the Norwegian Meteorological Institute (Hollingsworth, 1987), (Nordeng, 1986). Six-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. Baseline emissions of NO_x, SO_2 and NH_3 for Europe are taken from the 1990 EMEP inventory (Sandnes and Styve, 1992).

Regional scale modelling of ozone – the Source-Receptor Ozone Model
The EMEP MSC-W's Lagrangian Ozone model (Simpson, 1992, Simpson, 1993, Simpson, 1995) has been used to calculate the effects of reducing NO_x and VOC emissions from each country on the concentration levels for a number of countries in Europe, generating so-called source-receptor blame matrices (Simpson *et al.,* 1997). The regional modelling of ground-level ozone in EcoSense (Source-Receptor Ozone Model (SROM)) is based on an iteration model from Simpson and Eliassen (1997) which builds on these matrices.

Based on annual emissions of NO_x and NMVOC, the model calculates O_3 annual mean concentrations as well as concentration indicators such as Accumulated Ozone above a Threshold of 40 ppbV (AOT40) for the growing seasons of crops and forests and Accumulated Ozone above a Threshold of 60 ppbV (AOT60). The maximal spatial resolution is restricted by the applied country-to-grid blame matrices. On the source side the maximal resolution is country-level and on the concentration side about 150 by 150 km^2. The applied EMEP 150 grid has the same orientation as the EMEP 50 grid. Each of its grid cells is composed of nine EMEP 50 grid cells.

In order to be able to take non-linearities in the processes of ozone formation into account, two background levels are considered, the 1990 background (Base0) and a reduction in NO_x and NMVOC emissions by 70 percent (Base1). Outgoing from those levels, matrices for increments (Base1) and decrements (Base0) of 20 and 40 percent for NO_x and 40 percent for NMVOC emissions were estimated by Simpson *et al*. (1997) and implemented into SROM.

5.2 Multi-Compartment (air/water/soil) Analysis

In past work of the ExternE project series on external costs of energy, exposures and resulting impacts through contaminants present only in air were assessed and valued. In order to perform the external cost assessment in as complete a way as possible, the assessment has recently been broadened also to comprise exposures through food and drinking water. This requires models that also take the media soil and water as well as food items into account. In contrast to the assessment of purely airborne pollutants, these models do not only need to consider the environmental fate of a substance, i.e., its dispersion and transformation in the environment, but also the exposure particularly of human beings (when assessing human health impacts).

As there are some hundreds or even thousands of substances that may be hazardous, a prioritisation of the substances to be initially assessed was made. As a result, toxic substances that are released from power plants should be considered. Of particular concern are the toxic metals As, Cd, Cr, Hg, Ni and Pb (e.g., French *et al*. 1998; United Nations - Economic Commission for Europe 1998) which were consequently selected for study.

Available models were reviewed (e.g., European Commission, 1996; United States - Environmental Protection Agency, 1998; Huijbregts *et al*., 2000; Hertwich *et al*., 2001; International Atomic Energy Agency, 2001; McKone and Hertwich, 2001; McKone and Enoch, 2002; Pennington *et al*., 2005) with the conclusion that none of these models can be used directly for the calculation of external costs. This is because they do not quantify the total impact of an emitted pollutant but only the impact in a limited region, over a limited time horizon or on a limited population (the most exposed subgroup). Since the external cost should take into account the total impact (expectation value rather than worst case estimate) over all time, all space and the entire population, these models have to be adapted. However, by suitable modifications and adaptations two independent models have been developed and applied. One of the models (the "Uniform World Model") is based on transfer factors and other parameters of United States - Environmental Protection Agency (1998), the other ("WATSON") is a multi-zonal model that links the regional air quality model of EcoSense (cf. European Commission 1999) to a soil and water multimedia model of the Mackay level type (cf. Mackay, 2001). The output of these models is the damage per kg of pollutant or per kWh, as a function of the site and conditions (for emissions to air: stack height, exhaust temperature and velocity) of the source.

The goal of this section is to describe these two models that have been developed for the purpose of assessing a contaminant in the environment and also its exposure via food and drinking water to humans.

5.2.1 Uniform World Model

The starting point is the observation that, for incremental impacts due to small (compared to background levels) changes in emissions, the dose-response function (DRF) can be linearised and the corresponding total damage can be calculated with equilibrium models (steady state) even though the environment is never in equilibrium.[6] The necessary equations and parameters for the assessment of As, Cd, Cr, Hg, Ni and Pb are obtained from United States - Environmental Protection Agency (1998). The model is a generalisation to multimedia of the "uniform world model" for air pollution of Curtiss and Rabl (1996) and Spadaro (1999); it provides typical results for a region rather than for a specific site. Nonetheless it can distinguish, by means of simple correction factors, different kinds of sources such as power plants, industrial boilers and cars.

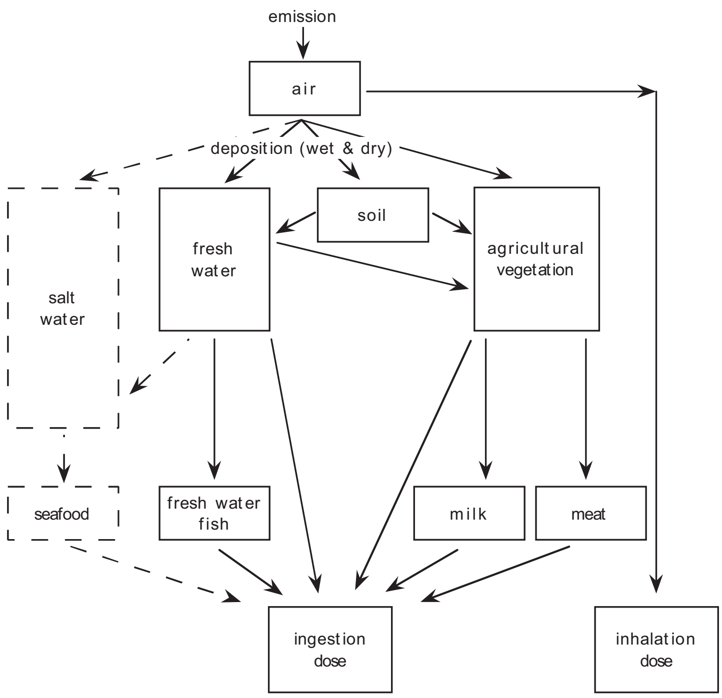

Figure 5.4 Pathways taken into account for health impacts of air pollutants by the Uniform World Model. Direct emissions to soil or water are a special case where the analysis begins at the respective "soil" and "water" boxes. In the present version seafood is not yet included.

We account for the pathways in Figure 5.4. We do not consider dermal contact because that pathway has been found to be entirely negligible for these metals (e.g. United States - Environmental Protection Agency, 1998; McKone and Enoch, 2002). Like the underlying model of United States - Environmental Protection Agency (1998), we do

[6] However, since some processes for toxic metals involve very long time constants τ, we also perform calculations where such processes are truncated with cut-off times of 30 and 100 years; for that we reduce the concentrations in the corresponding compartments by a factor $1\text{-}\exp(-t_{cutoff}/\tau)$.

not consider ground water, assuming that on average inflow and outflow of the pollutant to this compartment are equal. In the same spirit we assume that all drinking water is taken from surface water rather than groundwater. The resulting drinking water dose is an upper bound because it does not account for removal processes during the passage to and from groundwater.

We do not yet have all the elements for calculating the dose due to ingestion of seafood, which is potentially large because of bioconcentration and because most fish comes from the ocean rather than freshwater. One would need compartment models of all the oceans, coupled with data on fish production. Even if the concentration increment in the sea is very small, the collective dose from seafood could be significant if the removal processes (sedimentation) are slow and the analysis has no cut-off in time. The problem of long time constants also haunts the assessment of pathways that pass through soil. Neither United States Environment Protection Agency (EPA) nor International Atomic Energy Agency (IAEA) consider the impacts beyond the lifetime of the emitting installation, typically a few decades. When concerned with total impacts, two sets of results are regularly computed: one for the totality of the collective dose, and one for the collective dose incurred during the first 100 years. To allow valuation of the costs beyond the first generation with a lower intergenerational discount rate, the fraction of the dose incurred during the first 30 years after an emission is regularly indicated. The model is fully documented in Spadaro and Rabl (2004).

5.2.2 WATSON

The second model proposed covers the whole of Europe in a spatially-resolved way. It is called the *integrated **WAT**er and **SO**il environmental fate, exposure and impact assessment model of **N**oxious substances* (WATSON) *for Europe* and is coupled to the software tool EcoSense.

In order to allow for a bottom-up impact assessment approach that is in agreement with the impact pathway approach of ExternE, the media soil and water need to be modelled in a more spatially-resolved way for the whole of Europe. Compared to air, however, water and especially soils show highly variable properties so that there is quite a substantial literature on the most appropriate spatial and also temporal resolution at which these media would best be modelled (e.g. Addiscott, 1998; Becker, 1995; Hoosbeek and Bryant, 1992; Kirkby *et al.,* 1996; Blöschl, 1996). Models that cover larger areas than just a catchment with a fair degree of spatial resolution usually operate on a grid and most often cover the whole globe as global (atmospheric) circulation models. However, their focus is on the water balance or global biogeochemical cycles rather than on toxic substances. Although the modelling based on lumped parameters at larger scales is seen very critically (Becker, 1995), the model to be developed also needs to be acceptable in terms of computing time and data storage needs as it is meant to be a decision-support tool rather than purely serving research purposes. This is supported by Addiscott (1993) who pointed out that functional models are likely to be increasingly advantageous also with respect to their performance when the physical scale of the modelling exercise increases.

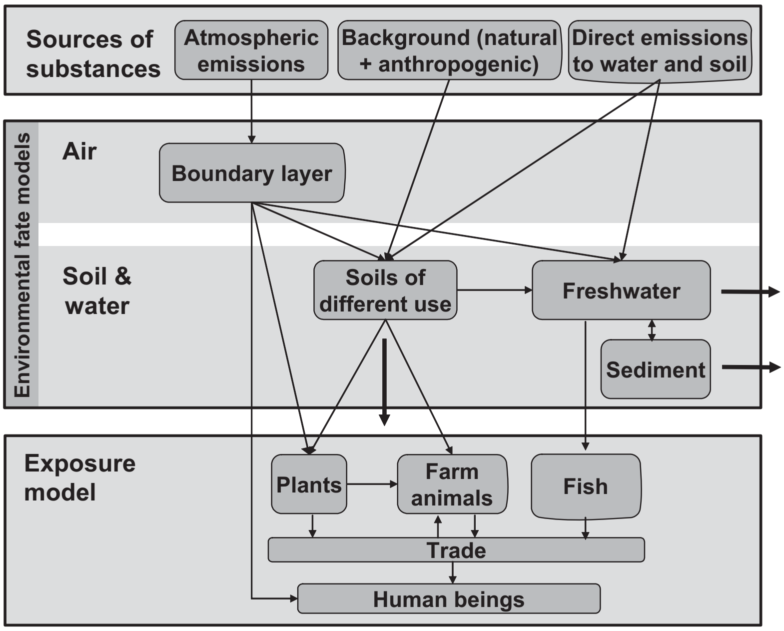

Figure 5.5 Conceptual structure of the environmental fate and exposure assessment of the WATSON model and its linkage to the air quality model contained in the EcoSense tool (arrows connecting boxes denote a substance's environmental pathway; arrows not connecting boxes indicate ultimate removal processes from the model's scope)

As a consequence, the multimedia modelling approach according to Mackay (2001) has been followed here which is well suited to quantitatively assess average concentrations of rather persistent substances at the regional scale resulting from highly dispersed and diffused sources (Cowan *et al.,* 1995). It is based on a mass balance that is formulated as a set of linear first-order ordinary differential equations. In line with Brandes *et al.* (1996), the mass balance is formulated based on concentrations. With the help of Mackay-type models, usually the steady-state solution is computed which assesses the situation when no mass change in any modelled compartment occurs due to continuous release of a substance over longer time periods. The time period until such a steady-state is reached actually depends on the nature of the substance, particularly its overall persistence in the modelled environment. Therefore, WATSON offers the opportunity not only to calculate a substance's environmental concentration in water and soil at steady state (which may serve as an indicator for sustainability if compared to a societal target value) but also dynamically with variable time steps. In addition, the time to reach a specified percentage of the steady-state concentration can be computed in order to get an impression of what time scales one has to deal with under a certain emission scenario until this ultimate situation occurs. Unlike many existing multimedia models, WATSON offers the option to switch particular processes on and off according to the nature of the substance rather than setting parameters to unreasonable values (e.g. for vapour pressure of metals other than Hg in Guinée *et al.,* 1996) since different processes are of varying importance for different substances. The processes that are covered by WATSON can be divided into different types (Table 5.1 also gives the processes considered).

Table 5.1 Process types and related processes considered in WATSON.

Process type	Processes
Transformation[a]	• degradation, • decay
Exchange	
inter-zonal	• river discharge • circulation of large lakes[b]
intra-zonal	Terrestrial environment: • matrix leaching, • preferential flow, • soil erosion, • overland flow and interflow, • ice melt of glaciers, • harvest removal Aquatic environment: • sedimentation, • re-suspension, • sediment burial, • diffusive exchange between water and sediments, • removal via catch of fish
Direct and diffuse input	• dry and wet atmospheric deposition • direct releases into water and soil

[a] irrelevant for the trace elements considered
[b] if a lake is fully contained in a zone it is already assumed to be fully mixed or homogeneous as part of a freshwater compartment according to multimedia modelling practice.

One drawback for coupling an air quality model to a multimedia (soil and water) model could be that it is not fully integrated. This means that the assumed/expected multiple intermedia exchanges between air on the one hand and soil, water and/or vegetation on the other of the so-called multimedia organic pollutants for instance may not be warranted. For the bulk of substances which are not true 'multi-hop pollutants' (Klepper and den Hollander, 1999), however, the intermedia exchange (or feedback) is assessed to be small (Margni *et al.,* 2004). Heavy metals can principally enter the atmosphere via volatilisation and re-suspension when attached to particles. Apart from mercury, heavy metals do not have a significant vapour pressure so that volatilisation can be neglected. Suzuki *et al*. (2000) investigated the influence of wind erosion on the fate of rather persistent organic chemicals with the help of a (fully integrated) multimedia model. In a sensitivity analysis, they found that this process is negligible. Therefore, it is assumed here that also for (persistent) heavy metals this process can be neglected supported by the fact that it mainly occurs on plains in arid to semi-arid climates with little to no vegetation cover (Scheffer and Schachtschabel, 1989) which are not widespread in Europe.

It is, therefore, concluded that the coupling of a single-medium air quality model to a water and soil multimedia type of model is a valid approach for assessing average environmental concentrations of non-'multi-feedback' pollutants at the regional scale.

Environmental fate modelling

As already outlined above, the environmental fate model consists of an existing single-medium air quality model (the Windrose Trajectory Model WTM) linked to a water and soil multimedia type of model ('air model' and 'water and soil model' blocks in Figure 5.5). The multimedia soil and water environmental fate model divides Europe into about 3400 zones (see Figure 5.6) according to the HYDRO1k GIS dataset for basins (EROS Data Centre, 1996; for comparison: the air quality model WTM is based on the EMEP 50 x 50 km^2 grid with 6600 terrestrial grid cells in Europe). This dataset was derived from a digital elevation model on a 1 km^2 raster. Although it contains some deviations from the real water pathways over the land surface, it allows a complete division of Europe into drainage basins. Deviations that had been detected and considered severe by comparison to the European rivers and catchments database (ERICA Version, 1998, European Environment Agency Data Service, 1998) as well as to the Britannica Atlas (Cleveland *et al.,* 1984) were corrected. Each drainage basin generically consists of different compartments, i.e. soils of different land use (i.e., pastures, arable land, non-vegetated areas (e.g. rocks, open cast mining), semi-natural ecosystems (e.g. forests, heathlands), built-up areas, glaciers) and surface water bodies with corresponding sediments. Spatially-resolved information on watersheds, land use, pH and organic carbon content of soils as well as on hydrology were taken from several, mostly publicly available sources (cf. Global Land Cover Facility, 1996; New *et al.,* 1999; European Environment Agency, 2000; Global Soil Data Task, 2000; Lehner and Döll, 2001; Batjes, 2002; Döll *et al.,* 2003). Dependence of the partitioning coefficients on pH is included as it is regarded as the single most important parameter of the partitioning of metals which should at least be considered in human health risk assessments of metals (Sauvé *et al.,* 2000).

Similar to the Universal World Model described above, no seawater compartment and corresponding sediment are included at present. Due to marine currents and migrating animals, there would be a need to model the entire oceanic system on Earth for long-lived (toxic) substances which in turn are the substances of highest concern. As a consequence, the modelling framework is as yet not capable of estimating the exposure due to marine fish consumption which to rather high degrees contributes to exposure to e.g. methyl-mercury or dioxins (e.g. French *et al.,* 1998; Buckley-Golder, 1999; Anonymous, 2000).

Innovations towards existing multimedia environmental fate models particularly take account of the rather persistent nature of the trace elements investigated. In contrast to organic substances for which particularly the degradation half life in the respective media is crucial, processes other than chemical transformations that contribute to the removal of the trace elements out of a compartment had initially been expected and later confirmed to be most important when assessing human exposure towards these contaminants. The respective innovations realised are:

- consideration of preferential flow: this is a process that takes into account that substances in the water phase of soils are not necessarily in equilibrium with the matrix; these may therefore be preferentially transported to the subsurface;
- compartment-specific soil erosion rates: different land uses show different resistances towards water soil erosion (cf. crop management factor; Golubev, 1982); this means that persistent substances reside longer in permanently vegetated pastures for instance than in arable soils with changing crops that show different degrees of soil cover;
- distinction of streams from lakes as regards their particle dynamics: ordinary multimedia models assume that any freshwater body at the land surface behaves like a lake disregarding that, under rapidly flowing conditions, the removal of substances from the water column is mostly driven by water flow rather than by sedimentation of particles.

Exposure modelling

The predicted environmental concentrations from the environmental fate module are used to assess the exposure to living organisms and finally to humans (Figure 5.6 ‘exposure model’ bar). There are basically three routes of exposure towards environmental chemicals which may lead to an impact: inhalation, ingestion and/or dermal contact. For inhalation, a combined exposure and impact assessment approach is followed by using exposure-response functions as have been widely applied in the series of ExternE projects (Friedrich and Bickel, 2001; European Commission, 1999).

Besides direct exposure via inhalation, the main indirect exposure route is ingestion of food and drinking water; dermal exposure as the third main route of exposure was left out in this investigation as this route of exposure to environmental pollutants is of much less concern compared to occupational exposure and exposure via cosmetic products. Modelling drinking water exposure for all European residents is a task that nobody has until now addressed following a detailed site-dependent bottom-up approach that aims at giving best estimates rather than those based on conservative (reasonable) worst-case scenarios. This is because it is groundwater that constitutes a major part of the drinking water resources (Scheidleder *et al.,* 1999). Even at smaller scales there is a failure to model mass transfers in groundwater aquifers due to lack of information (e.g. Eggleston and Rojstaczer, 2000). It also appears that groundwater contamination due to heavy metals for instance is a very localised problem and in the case of heavy metals is confined to areas with former or present mining activities (Stanners and Bourdeau, 1995). Due to the lack of contamination as well as aquifer information, a modelling effort would at present result in rather unreliable concentration estimates. Thus, exposure via drinking water is for the moment not included in the proposed modelling framework.

The assessment of the exposure via food ingestion is more complex than that via inhalation. This is because different food chains need to be taken into account. A fairly simple food chain, for instance, is a plant that is eaten by a cow whose meat in turn is eaten by human beings. A toxic substance that comes with the plant – the substance may actually have been taken up via roots or leaves or may just adhere to plant parts – is distributed between milk, meat, inner organs, or the excrements or urine of the cow. The situation becomes even more complex when dealing with wild animals and especially with fish due to the unmanaged food supply. After ingestion by humans again a distribution between different body parts takes place of which only some locations are prone to damages by the substance (WHO, 2000).

Since in the present study we focus on heavy metals, the exposure assessment of United States - Environmental Protection Agency (1998) has been followed similar to UWM. A further restriction is that not all food-ingestion-related exposure pathways are included in WATSON at present. In particular exposure via seafood is not considered. As was argued above, modelling the marine environment almost inevitably brings about the necessity to extend the geographical scope of the model to the whole globe. Thus, seafood consumption is as yet not included. Although the exposure assessment due to ingestion is not exhaustive, exposure via staple food products are to a large degree considered (i.e., wheat, barley, rye, potato, spinach, beef, cow milk and products, pork, poultry, eggs).

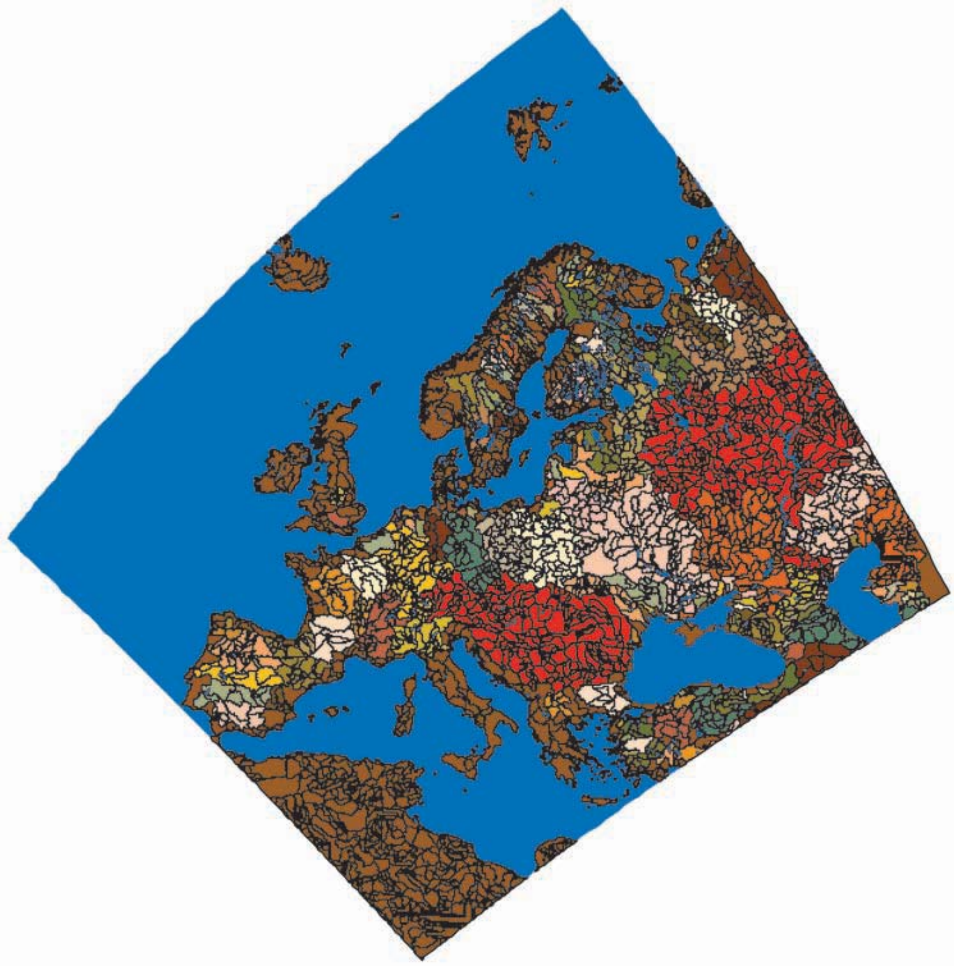

Figure 5.6 Geographical scope of WATSON-Europe corresponding to the receptor area of the EcoSense model. Catchments consisting of more than one region are coloured (derived on the basis of EROS data centre, 1996)

Unlike inhalation and exposure via drinking water, the exposure via food does not only need to take into account the environmental concentration and the transfer into plants and/or animals but also the trade of food that contains a substance which causes an adverse effect. Only the trade within Europe is considered. For this it is assumed that the food items are equally distributed over the whole European/Asian receptor area of WATSON (see Figure 5.6) weighted by the stocks or the produced amounts of livestock and crops, respectively. These are taken from the data already implemented in EcoSense and supplemented by data taken from FAO statistical database (FAOSTAT, 2002). This approach is of course a generalisation of the real path of food products or, on the other hand, of the actual exposure scenario. It is very different from typical risk assessment frameworks where the conservative ‘subsistence farmer exposure’ scenario is often used (European Commission, 1996). Allowing for trade rather is in line with Pennington *et al*. (2005) who introduced a ‘production-based’ approach where a so-called intake fraction (e.g. Bennett *et al.,* 2002) assesses the portion of an emission to which a population will be finally exposed. The intake fraction is, thus, a good measure on which to base exposure-response functions in order to get representative impact estimates. It is used by WATSON as an indicator of population exposure. The WATSON model is fully documented in Bachmann (2006).

5.3 Sound Propagation

Noise is unwanted or damaging sound. It is emitted from almost all stages of all fuel cycles. 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.

Transport noise has been recognised as a very important issue for a long time. For this reason, it is well studied and propagation models are available for road, rail and air transport. 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.

The propagation of sound through the atmosphere is well understood at the theoretical level. Nevertheless, modelling the propagation in concrete applications presents some practical problems, mainly due to the amount of detailed input data required.

The EC Directive on Environmental Noise and its requirement for member states to prepare noise maps has increased interest in standardising noise propagation models across the EU (see e.g. European Commission, 2003).

Quantification of transport noise impacts with the ExternE methodology has been based on two German semi-empirical standard models. Road noise is modelled using *RLS90* (Richtlinien für den Lärmschutz an Strassen, see Arbeitsausschuß Immissionschutz an Strassen, 1990). The model was enhanced to allow the use of more than two vehicle categories and the respective emission functions, as well as individual vehicle speeds per category. Noise propagation for rail transport is modelled according to the German rail noise model *Schall03* (Bundesbahn, 1990). For the calculation of impacts, different noise indices are calculated: $L_{Aeq(7.00\text{-}19.00)}$, $L_{Aeq(19.00\text{-}23.00)}$, $L_{Aeq(23.00\text{-}7.00)}$ and L_{DEN} (composite indicator). Noise levels are calculated as incident sound at the façade of the buildings. More information can be found in Bickel *et al*. (2003).

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6 Impact Pathway Approach: Exposure-Response Functions

6.1 Overview of Health Impacts of Air Pollution

Health impacts are especially important for ExternE because in terms of costs they contribute by far the largest part of the total, apart from global warming. A consensus has been emerging among public health experts that air pollution, even at current ambient levels, aggravates morbidity (especially respiratory and cardiovascular diseases) and leads to premature mortality (e.g. Wilson and Spengler, 1996; WHO, 2003; Holland *et al*., 2005a). There is less certainty about specific causes but most recent studies have identified fine particles as a prime culprit; ozone has also been implicated directly. The largest contribution to the damage cost comes from mortality due to particulate matter (PM). Another important contribution arises from chronic bronchitis due to particles (Abbey *et al*., 1995). In addition there may be significant direct health impacts of SO_2, but for direct impacts of NO_x the evidence is less convincing.

Table 6.1 Air pollutants and their effects on health.

Primary Pollutants	Secondary Pollutants	Impacts
Particles (PM_{10}, $PM_{2.5}$, black smoke)		mortality cardio-pulmonary morbidity (cerebrovascular hospital admissions, congestive heart failure, chronic bronchitis, chronic cough in children, lower respiratory symptoms, cough in asthmatics)
SO_2		mortality cardio-pulmonary morbidity (hospitalisation, consultation of doctor, asthma, sick leave, restricted activity)
SO_2	Sulphates	like particles?
NO_x		morbidity?
NO_x	Nitrates	like particles?
NO_x+VOC	Ozone	mortality morbidity (respiratory hospital admissions, restricted activity days, asthma attacks, symptom days)
CO		mortality (congestive heart failure) morbidity (cardio-vascular)
PAH diesel soot, benzene, 1,3-butadiene, dioxins		cancers
As, Cd, Cr-VI, Ni		cancers other morbidity
Hg, Pb		morbidity (neurotoxic)

The reason for the question marks in the lines for sulphates and nitrates is the lack of specific evidence for their toxicity. They constitute a large percentage of ambient PM, but most of the available epidemiological studies are based simply on the mass of PM without any distinction of the components or characteristics (acidity, solubility, etc.). In particular there is a lack of epidemiological studies of nitrate aerosols because, until recently, this pollutant was not monitored by air pollution monitoring stations.

Quite generally it is difficult for epidemiologists to attribute a particular health impact to a particular pollutant, because populations are exposed to a mix of different pollutants that tend to be highly correlated with each other. The conclusion that air pollution damages health is much more certain than the attribution of damage to a particular pollutant. For that reason some epidemiologists, especially in France, keep emphasising that any individual pollutant is merely an indicator of pollution and that the attribution of an impact to a specific pollutant is very uncertain (ERPURS, 1997). American epidemiologists have tended to attribute the damage mostly to PM, although in recent years they have also recognised the possibility of a larger role of the other pollutants.

6.2 Dose-response Functions and the Calculation of Impacts

6.2.1 Form of the dose-response function

The dose-response function (DRF) relates the quantity of a pollutant that affects a receptor (e.g. population) to the physical impact on this receptor (e.g. incremental number of hospitalisations). In the narrow sense of the term, it should be based on the dose actually absorbed by a receptor. However, the term dose-response function is often used in a wider sense where it is formulated directly in terms of the concentration of a pollutant in the ambient air, accounting implicitly for the absorption of the pollutant from the air into the body. The functions for the classical air pollutants (NO_x, SO_2, O_3, and particulates) are typically of that kind and the terms exposure-response function (ERF) or concentration-response function (CRF) are often used.

The DRF is a central ingredient in the impact pathway analysis and merits special attention. A damage can be quantified only if the corresponding DRF is known. Such functions are available for the impacts on human health, building materials and crops, caused by a range of pollutants such as primary and secondary (i.e. nitrates, sulphates) particles, ozone, CO, SO_2, NO_x, benzene, dioxins, As, Cd, Cr, Ni and Pb. The most comprehensive reference for health impacts is the IRIS database of the US EPA (http://www.epa.gov/iriswebp/iris/index.html). For the application in an impact pathway analysis, that information often has to be expressed in a somewhat different form, accounting for additional factors such as the incidence rate (European Commission, 1999; Spadaro and Rabl, 2004).

Unfortunately, for many pollutants and many impacts, the DRFs are very uncertain or not even known at all. For most substances and non-cancer impacts, the only available information covers thresholds, typically the NOAEL (no observed adverse effect level) or LOAEL (lowest observed adverse effect level). Knowledge of thresholds is not sufficient for quantifying impacts; it only provides an answer to the question whether or not there is a risk. The principal exceptions are carcinogens and the classical air pollutants, for which explicit DRFs are known (often on the assumption of linearity and no threshold). Recently Pennington *et al.* (2002) have proposed a promising method of using LOAEL or NOAEL data for estimating DRFs, but their results are not yet sufficiently complete for use by ExternE.

By definition a DRF starts at the origin and in most cases it increases monotonically with dose, as sketched schematically in Figure 6.1. At very high doses the function may level off in an S-shaped fashion due to saturation, but that case is not of interest here. DRFs 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.

A major difficulty lies in the fact that one needs relatively high doses in order to obtain observable non-zero responses unless the sample is very large; such doses are usually far in excess of typical exposures in the EU or North America. Thus there is a serious problem of how to extrapolate from the observed data to low doses. Figure 6.1 indicates several possibilities for the case where the point P corresponds to the lowest dose at which a response has been measured. The simplest is the linear model, i.e. a straight line from the origin through the observed data point(s). The available evidence suggests that a dose-response function is unlikely to go above this straight line in the low dose limit. The straight line model does appear to be appropriate in many cases, in particular for many cancers.

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.

There is even the possibility of a "fertiliser effect" at low doses, as indicated by the dashed line in Table 6.1. This can be observed, for example, in the dose-response functions for the impact of NO_x and SO_2 on crops: a low dose of these pollutants can increase the crop yield, in other words the damage is negative. Generally a fertiliser effect can occur with pollutants that provide trace elements needed by an organism. Cr is an interesting example to illustrate the complexity of possible effects: as an essential element for the human body it is beneficial at low doses, but at high dose it is toxic; in addition it is carcinogenic if in oxidation state VI.

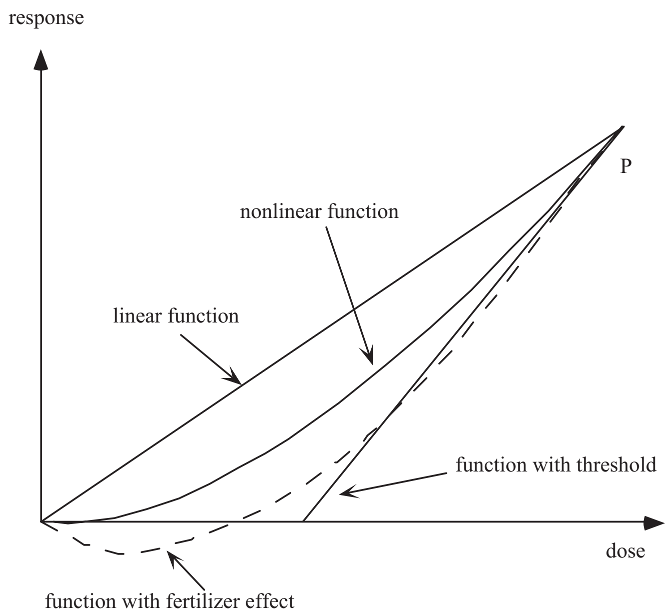

Figure 6.1 Possible behaviour of dose-response functions at low doses: the four functions shown have the same value at P (the lowest dose where a non-zero impact has been observed).

The CRFs for NO_x, SO_2, PM and O_3 that have been or are used by ExternE are assumed to be linear without threshold. In support of linearity, one can cite the following:

- Fig. 3 of Dockery *et al*. (1993);
- Figs. 1 and 2 of Pope *et al*. (1995);
- The graph on p.64 of ERPURS (1997) for SO_2;
- The meta-analysis of 107 studies by Zmirou *et al.* (1997) which states in its English abstract "The dose-response functions seem linear in the range of observed concentrations" for PM, O_3, SO_2 and NO_2;
- The study of air pollution mortality in nine cities in France by Zeghnoun *et al.* (2001) which states in its English abstract "These associations were linear without threshold";
- The consistency of CRF slopes across a wide range of concentrations;
- The review by Daniels *et al.* (2000) who conclude that "… linear models without a threshold are appropriate for assessing the effect of particulate air pollution on daily mortality even at current levels".

In contrast to the homogeneous populations of cloned animals studied by toxicologists, the absence of a no-effect threshold for population-level DRFs is plausible because real populations always contain individuals with widely differing sensitivities; for example, at any moment about 1% of a population with a 75 year life expectancy is within the last nine months of life and thus extremely frail. Thus the question of thresholds depends on what type of population it is based. DRFs for individuals or for a group of fairly similar individuals are likely to have a no-effect threshold: for example in a group of young and healthy individuals a moderate air pollution peak will not induce any premature deaths. The DRFs used by health impact assessments and by ExternE are based on entire populations.

For cancers most authors assume linearity without threshold. That is the most plausible model on theoretical grounds for substances that initiate a cancer. For substances that promote the growth of a cancer the DRF could have a no-effect threshold but there are few specific data. A very interesting study of this issue is the one of Frith *et al.* (1981) who, using a very large number of mice, were able to measure the DRFs of a certain carcinogen associated with two types of cancer: for one the DRF has a threshold; for the other it is linear without threshold.

For the particles, NO_x, SO_2, O_3 and CO, the background in most industrialised countries is above the level where effects are known to occur. However, when evaluating large reductions of exposure to levels below those that have been observed in epidemiological studies, the possibility of a no-effect threshold cannot be ruled out and the uncertainties are large. In particular, some experts prefer to assume a no-effect threshold for O_3; for example the Task Force on Health of WHO-UNECE assumed a threshold of 35 ppb (70 $\mu g/m^3$) for daily maximum 8-hour mean O_3 concentration.

Note that, 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, only the slope matters. Since a straight line through the origin is uniquely characterised by its slope, we state all CRFs in terms of their slope s_{CR}.

6.2.2 Difficulties of epidemiological studies

CRFs for air pollution are determined by epidemiological studies using statistical analysis. The correlations between pollution and a health impact (called an end-point) are called associations. The uncertainties of any single study are very large for several reasons. First of all, the health impacts are small at typical concentrations – fortunately for us: we are not all dropping dead from pollution. By the same token, it is difficult for epidemiologists to measure the impacts.

Secondly, in contrast to the extreme complexity of the underlying biological processes, epidemiological studies can take into account only certain simple gross features, for example the variation of respiratory hospital admissions as a function of the SO_2

concentration to which the study population is exposed. Even the most detailed studies cannot take into account more than a few limited characteristics of a situation.

For example, the most commonly used methodology, analysis of time series, looks for correlations between the daily frequency of an end-point and the daily concentration to which the population is exposed as measured by monitoring stations of the local air quality network. Individual differences are taken into account only to the extent that the health data can distinguish them, for example by distinguishing hospital admissions for patients over and under the age of 65. Even if the population includes large metropolitan areas, the number of cases per day is not sufficiently large to allow very fine distinctions between different groups of individuals (the smaller the number of cases per day, the more uncertain the attribution of variations to pollution).

A time series analysis can only identify acute impacts (short-term impacts), i.e. impacts occurring within a few days after exposure (in practice almost no such studies are able to take into account a lag between exposure and impact of more than five days). Measuring chronic impacts, due to chronic exposure, requires long-term observations that are much more difficult and costly. The terms acute and chronic are also applied to mortality even though they appear strange in that context: acute mortality is the mortality impact within a few days of exposure, chronic mortality is the total due to chronic exposure.

Thirdly, populations are exposed to a mix of different pollutants that tend to be highly correlated with each other. Therefore it is difficult to establish definite links between an end-point and a particular pollutant. Many studies look at several air pollutants, comparing the results of different regressions, but very few are able to provide regression models with several pollutants and rarely more than two.

The reason for the focus on PM is that the most consistent results world-wide have been found for PM (a possible explanation is that the indoor concentration is close to the one measured outdoors at monitoring stations, in contrast to most of the gaseous pollutants for which the indoor/outdoor ratio is highly variable). In particular, multipollutant analyses have usually found PM to be the most significant. This is despite the fact that PM is an ill-defined mixture of pollutants (anything, solid or liquid, that accumulates in a particle detector, including sulphuric acid and ammonium nitrate) whose composition can be quite different at different sites. Studies on the relative toxicity of different components of PM have, so far, not been sufficiently conclusive to draw firm conclusions, although sulphates do appear in quite a few significant associations, in particular in Pope *et al.* (1995). For example, some studies find associations with acidity, but others do not (Lippman *et al*., 2000). Several recent studies have found that crustal particles, a major constituent (typically 10 to 50%) of ambient PM_{10}, appear to be harmless (Laden *et al*., 2000; Pope *et al*., 1999; Schwartz *et al*., 1999); most of the damage seems to be caused by combustion particles.

The question of how to analyse the contribution of a particular pollutant in the mixture will be discussed in section 6.2.5.

6.2.3 Differences between results of different studies

As discussed above, CRFs obtained from different populations might reflect different sensitivities but they could also be due to differences in the local pollution mix, to say nothing of differences in methodology (for example the choice of the time lag between concentration and end-point in a time series analysis).

It is not surprising that different studies find different results; some, for instance, find strong effects of SO_2 while others do not. In some cases a reanalysis of the same data with improved methodology has obtained results that are appreciably different. For example, when the results of the APHEA Project were published in 1997, it seemed that the acute mortality CRF for PM_{10} in Europe had only about half the slope of the average found by numerous studies in the North America. The latter had mostly been done for single locations and not always following the same protocol. More recently an improved and more comprehensive analysis of pooled data in the USA (HEI, 2001) obtained essentially the same slope as APHEA which had used a standardised protocol for 15 cities. To cite from HEI, *"...HEI's US-wide National Morbidity, Mortality and Air Pollution Study (NMMAPS) (Samet et al., 2000) found a 0.5% increase in total non-accidental mortality associated with a 10 µg/m³ increase in PM_{10} in the 90 largest US cities where daily average PM_{10} ranged from 15 to 53 µg/m³. This result agrees closely with that of the European APHEA study (0.6% per 10 µg/m³)(Katsouyanni et al., 1997) and with a recent meta-analysis of 29 studies in 23 locations in Europe and North and South America (0.7% per 10 µg/m³) (Levy et al., 2000)."*

Another illustration of the danger of relying on a single study can be seen in the APHEA results for Eastern Europe: initially they seemed to imply a much smaller CRF slope than for Western Europe. However, a recent re-analysis (Samoli *et al.*, 2001) found *"... The ratio of western to central-eastern cities for estimates was reduced to 1.3 for BS (previously 4.8) and 2.6 for SO_2 (previously 4.4). We conclude that part of the heterogeneity in the estimates of air pollution effects between western and central-eastern cities reported in previous publications was caused by the statistical approach used and the inclusion of days with pollutant levels above 150 µg/m³ ...."*

Interesting results have been published recently that shed some light on the variability of CRFs between different regions. For example, s_{CR} for acute mortality varies between different regions of the USA, being more than twice as large in the North East than in the Southwest (with the exception of Southern California where it is almost as high as in the North East) (HEI, 2001). Table 6.2 summarises results on regional variability in Europe from the APHEA2 study (Katsouyanni, 2001). The range of variation is large and roughly comparable to HEI (2001). These results are not sufficient, however, to allow generalisation to other locations.

For these reasons we recommend, at this time, to use the results obtained in Europe and North America, rather than country-specific values that may be available from the very limited number of studies carried out elsewhere.

Table 6.2 Variation of acute mortality due to PM_{10} in Europe (Katsouyanni, 2001; in parentheses 95% CI).

	% increase per 10 $\mu g/m^3$
Average, Europe	0.60% (0.40-0.80%)
City with low average NO_2	0.19% (0.00-0.41%)
City with high average NO_2	0.80% (0.67-0.93%)
Cold climate	0.29% (0.16-0.42%)
Warm climate	0.82% (0.69-0.96%)
City with low standardised mortality rate	0.80% (0.65-0.95%)
City with high standardised mortality rate	0.43% (0.24-0.62%)

Recent reanalysis of the NMMAPS data with improved models (GAM and GLM) resulted in lower CRFs: about 0.21% to 0.27% per 10 $\mu g/m^3$ PM_{10} increase (HEI, 2003). However, the WHO meta-analysis of European studies recommends a CRF very close to the original estimates,

0.6% per 10 $\mu g/m^3$ for all-cause mortality, all ages, due to PM_{10}

and that value is now used for ExternE.

6.2.4 Relative risk and CRF

Most epidemiological studies report their results in terms of relative risk (RR), defined as the ratio of the incidence observed at two different exposure levels. (Case-control studies observe subgroups that are not representative of the entire population, and they report only an odds ratio (OR); in the limit of small effects, relevant for most air pollution impacts, the OR is approximately equal to the RR). In order to quantify damages one needs to translate RR in terms of a CRF for the incremental cases per exposure increment. The number of cases at a relative risk RR is the product of RR and the baseline or reference level of incidence I_{ref}. Most epidemiological studies do not provide data for I_{ref} and other sources must be consulted.

For the present purpose we find it convenient to define all CRFs in terms of cases per year per average person per $\mu g/m^3$, because then they can be applied directly to the entire population without worrying about affected subgroups. Therefore we also include a factor f_{pop} that is equal to the fraction of the population affected by the end-point in question.

Transferring CRFs to other countries requires data for the respective I_{ref} together with an assumption about the RR: is the RR the same for populations different from the one in the epidemiological study? There are few data to answer that question and the uncertainties can be large.

6.2.5 Which pollutant causes how much health damage?

Several health impact assessments, for example by the World Health Organisation (WHO, 2003), estimate the health impacts of exposure to ambient levels of air pollution. This is sufficient for informing policy makers about the benefit of reducing the concentration values recommended as guidelines for ambient air quality. For such an assessment the results of epidemiological studies can be used without any hypotheses about the toxicity of different components of ambient PM: both the studies and the assessments are based directly on typical compositions of ambient PM.

In contrast, ExternE is a bottom-up methodology and starts from the source of the pollutants, calculating the damage attributable to each emitted pollutant (called primary pollutant). The need for this kind of information becomes obvious when one recognises that, in order to actually attain lower ambient concentrations, specific regulations must be put in place to force polluters to reduce their emissions. For the optimal formulation of such regulations one needs to compare the benefits of reducing the emission of a pollutant and the cost of such a reduction for all abatement technologies under consideration. In some cases tradeoffs must be made between the reductions of different pollutants; for example certain automotive technologies reduce the emission of PM while increasing the emission of NO_x. Thus the optimal formulation of environmental policies requires more detailed information on the health effects of specific pollutants: one needs to know the incremental impact of an incremental kg of each pollutant that is emitted by a particular source such as a power plant or a car.

Separating out the roles of SO_2, NO_2 and PM_{10} is particularly problematic, given that they tend to vary together in most locations and studies. It is not clear to what extent the apparent effects of PM are in reality a reflection of effects of NO_2 or SO_2 or vice versa, or whether the presence of other pollutants affects the toxicity of PM. Thus there are uncertainties in applying CRFs in a situation where the ambient pollutant mixture is different from the one where the original epidemiological study was carried out.

The current position of ExternE is to use only CRFs for PM and O_3 but none for SO_2 or NO_x, a choice also made in other health impact assessments. However, the situation is not clear and opinions could change as further evidence comes to light. In particular, the Hong Kong intervention study showed a sustained benefit in mortality reductions following reductions in pollution involving mostly SO_2 (Hedley *et al*., 2002). There could indeed be significant direct effects of SO_2, contrary to the current position of ExternE.

To proceed it is convenient to write the incremental impact ΔI for a particular end-point as a sum of the contributions of the individual pollutants (each with CRF slope s_i and concentration increment Δc_i):

$$\Delta I = \sum s_i \, \Delta c_i \tag{6.1}$$

the unit of I is cases per year per average person. The Δc_i are calculated for each location where there is human population and the impacts are summed over all locations to obtain the total for the entire region that is affected.

For the previous ExternE reports (European Commission, 1999; ExternE, 2000) the assumption was made that the toxicity of all sulphates is equal to that of $PM_{2.5}$ and the toxicity of particulate nitrates equal to that of PM_{10}. This distinction between sulphates and nitrates was based only on size, noting that nitrates need other particles to condense on, whereas sulphates self-nucleate and are therefore smaller on average. The ratio of CRF slopes $s_{PM10}/s_{PM2.5}$ was taken as 0.6, because this is a typical value of the ratio of concentrations of $PM_{2.5}$ and PM_{10}. The size, composition and toxicity of primary PM emitted by different sources can be quite different; for example, automotive PM is almost entirely organic or carbonaceous whereas PM from coal combustion contains in addition a sizeable proportion of minerals. Particles from internal combustion engines are all $PM_{2.5}$, whereas those from power plants are larger (mostly PM_{10}, with some particles being even larger than 10 μm). Since the available emissions data are simply stated in terms of PM mass, the best one can do is distinguish different typical PM compositions according to their source. In European Commission (1999) and ExternE (2000), ExternE treated power plant emissions as being equivalent in toxicity to PM_{10} and vehicle emissions as equivalent to $PM_{2.5}$. In terms of the above equations one can summarise the assumptions for the health impact ΔI due to a concentration increment Δc_i as

$$\Delta I = s_{PM10}\, \Delta c_{PMpower} + s_{PM2.5}\, \Delta c_{PMtrans} + s_{PM2.5}\, \Delta c_{sulph} + s_{PM10}\, \Delta c_{nitr} + s_{O3}\, \Delta c_{O3} + s_{SO2}\, \Delta c_{SO2} + s_{CO}\, \Delta c_{CO} + \text{other} \tag{6.2}$$

where

$\Delta c_{PMpower}$ = concentration due to primary combustion PM from power plants,
$\Delta c_{PMtrans}$ = concentration due to primary combustion PM from transport, and
"other" = analogous terms for carcinogens such as benzene.

and the ratio of CRF slopes is

$$s_{PM10}/s_{PM2.5} = 0.6\,. \tag{6.3}$$

For the current version of ExternE, the assumptions about the toxicity of the different PM types have been changed after a careful review of the latest epidemiological and toxicological literature. Evidence has been accumulating to underline the high toxicity of combustion particles and especially of particles from internal combustion engines. For the secondary particles the evidence is less convincing. In particular for nitrates there is still not much evidence for harmful effects, whereas for sulphates quite a few studies, including the very important cohort study of Pope *et al*. (2002), do find associations. Therefore ExternE now treats

- nitrates as equivalent to 0.5 times the toxicity of PM_{10};
- sulphates as equivalent to PM_{10} (or 0.6 times $PM_{2.5}$);
- primary particles from power stations as equivalent to PM_{10};
- primary particles from vehicles as equivalent to 1.5 times the toxicity of $PM_{2.5}$.

Effects of O_3 are considered independent of PM and added, whereas direct of effects of CO, SO_2 or NO_x are not taken into account. In equation form this can be written for the ExternE results of 2004 as

$$\Delta I = s_{PM10}\, \Delta c_{PMpower} + 1.5\, s_{PM2.5}\, \Delta c_{PMtrans} + s_{PM10}\, \Delta c_{sulph} + 0.5\, s_{PM10}\, \Delta c_{nitr} + s_{O3}\, \Delta c_{O3}, \tag{6.4}$$

with $s_{PM10}/s_{PM2.5} = 0.6$.

6.3 CRFs for Mortality

6.3.1 Loss of life expectancy vs. number of deaths

In recent years, many studies have attempted to quantify the impacts of mortality due to air pollution (ORNL/RFF, 1994; Rowe *et al*., 1995; European Commission, 1999; ExternE, 2000; Levy *et al.,* 1999; Abt, 2000; Kuenzli *et al.,* 2000; and others). Whereas all studies before 1996 calculated a number of premature deaths and applied a value of a prevented fatality (VPF) to obtain a monetary value of these deaths, there has been a growing recognition in recent years that it is more meaningful to look at loss of life expectancy (LE) (see e.g. McMichael *et al*., 1999; Wilson and Crouch, 2001). In particular the ExternE project series has, since 1998, based the monetary valuation of air pollution mortality on the value of a life year (VOLY).

Since this is not yet universally accepted and some analysts, especially in the USA, continue to use number of deaths as impact indicator, we list the main reasons why the VPF approach is wrong for air pollution:

- it makes no sense to add the number of deaths due to different contributing causes (such as air pollution, smoking or lack of exercise) because one would end up with numbers far in excess of total mortality;
- the number of deaths fails to take into account a crucial aspect for the monetary valuation, namely the magnitude of the loss of life per death, very different between typical air pollution deaths and typical accidents;
- in contrast to primary causes of death (such as accidents), the total number of premature deaths attributable to air pollution is not observable;
- the method that has been used for calculating the number of deaths for cohort studies is wrong.

The first two are obvious, the third and fourth are explained in Rabl (2003) who examined what exactly has been measured in epidemiological studies of mortality. He shows that the studies of chronic mortality cannot distinguish whether the observed result is due to everybody losing a little LE or only some individuals losing much. Analogy with studies of smokers suggest that everybody's life is shortened to some extent by air pollution; in that case the VPF approach would have to say that every death is an air pollution death – not a very meaningful conclusion.

There are several reasons for the popularity of calculations of a number of premature deaths due to air pollution. One is the emotional impact: "deaths due to pollution" sounds more dramatic than "years of life lost". Another reason is that such calculations seem natural and plausible in view of the fact that epidemiological studies report an increase in mortality rates. The reason for the error of such calculations for chronic mortality is not entirely obvious, as explained by Rabl (2003), it lies in the neglect of dynamic effects (if a pollution peak causes some individuals to die now, ΔLE sooner than otherwise, the number of deaths during the ensuing period of ΔLE will be lower because people cannot die twice). As far as damage costs are concerned, the results of the VPF and of the VOLY approach are not radically different. In fact, one can get exactly the same result if one chooses the loss of LE per death appropriately. However, there is essentially no good basis for estimating the LE loss per air pollution death.

Quantifying the loss of LE directly avoids the need for estimating the LE loss per air pollution death and thus avoids these problems. Loss of LE is a meaningful and appropriate impact indicator for all risk factors, even those that are not observable as the cause of an individual death. In particular, it can be added across different risk factors (at least in the limit of small risks).

Number of deaths seems appropriate only for acute mortality and for infant mortality; however, for monetary valuation these impacts are problematic because the epidemiological studies provide no information on the LE loss per death for these end-points.

6.3.2 Studies of chronic mortality

It is very difficult and costly to measure the total impacts (short plus long-term) of air pollution and there are only few long-term studies available. However, in recent years, several important epidemiological studies have succeeded in measuring the long-term impacts of air pollution on mortality. Two of these (Dockery *et al*., 1993; Pope *et al*., 1995) have found positive correlations between exposure to particles and total mortality, while the third (Abbey *et al*., 1999) found a positive correlation with mortality for men but not for women. Confirmation of long-term mortality impacts has recently been provided by a study in the Netherlands (Hoek *et al*., 2002). These studies are called cohort studies because they analyse the survival of a cohort of individuals over a long period, at least several years, and correlate it with individual exposure to air pollution.

Because the chronic mortality studies are of crucial importance for the formulation of environmental policy, the US EPA requested a reanalysis by an independent team. This reanalysis was carried out by Krewski *et al*. (2000); it confirmed the validity of the data and of the analysis of the original studies (Dockery *et al*., 1993; Pope *et al*., 1995). In addition, Krewski *et al*. performed a large number of sensitivity studies. In the meantime the cohort study of Pope *et al*. (1995) has continued and new results have been published by Pope *et al*. (2002), based on an observation period of about 16 years. Of the long-term mortality studies, the one by Pope *et al.* has by far the largest sample, about half a million individuals, and we use its results for ExternE.

Pope *et al.,* (2002) report two different numbers, 1.04 and 1.06, for the relative risk due to a 10 $\mu g/m^3$ increment of $PM_{2.5}$, depending on assumptions about the relevant exposure period. Here we use 1.05 as the average of these two values.

$$RR = 1.05 \text{ for a } 10\ \mu g/m^3 \text{ increment of } PM_{2.5}\ . \qquad (6.5)$$

Pope *et al*. (2002) also find correlations of the relative risk RR with the concentration of sulphates.

6.3.3 Loss of life expectancy for chronic mortality of adults

Since this relative risk refers to an increase in age-specific mortality, a more elaborate calculation involving life table data is required to find the corresponding YOLL (years of life lost). European Commission (1999) and Rabl (1998) report a relatively simple steady state calculation. A more detailed dynamic analysis was carried out by ExternE (2000) with a similar result. The most comprehensive dynamic analysis was published by Leksell and Rabl (2001) who evaluated the sensitivity of the result to the underlying assumptions.

Here we explain the steady state calculation. A key element for the analysis is the age-specific mortality $\mu(x)$, defined such that someone who has reached age x has a probability $\mu(x)\,\Delta x$ of dying between x and x + Δx (data are usually stated in terms of Δx = 1 year). The survival function S(x,x') is the fraction of a cohort of age x that survives at least to age x'. Since the fraction that dies between x' and x' + Δx' is $\Delta S_\mu(x,x') = S_\mu(x,x')\,\mu(x)\,\Delta x$, one gets the differential equation

$$dS_\mu(x,x') = - S_\mu(x,x')\,\mu(x')\,dx'; \qquad (6.6)$$

the boundary condition is $S_\mu(x,x) = 1$. One readily finds the solution

$$S_\mu(x,x') = \exp[- \int_x^{x'} \mu(x'')\,dx'']. \qquad (6.7)$$

The remaining life expectancy L(x) of a cohort of age x is

$$L(x) = \int_x^{\infty} S_\mu(x,x')\,dx'. \qquad (6.8)$$

If $\mu(x)$ changes, for example due to air pollution, $S_\mu(x,x')$ and L(x) change accordingly. The resulting change $\Delta L(x)$ for a cohort of age x is the difference between L(x) calculated without and with this increase

$$\Delta L(x) = \int_x^{\infty} [S_{\mu 0}(x,x') - S_\mu(x,x')]\,dx', \qquad (6.9)$$

where $S_{\mu 0}(x,x')$ is the survival curve for the baseline mortality $\mu_0(x)$. The impact on the entire population is obtained by summing $\Delta L(x)$ over all affected cohorts, weighted by the age distribution. Only ages above 30 have been included in the calculations because the underlying cohort studies did not include younger people. However, that is essentially no limitation as far as adult mortality is concerned because the relative risk found by Pope *et al*. seems to be independent of age, as shown by the data in Table 21 of the re-analysis by Krewski *et al*. (2000). Since the absolute mortality is very low between the end of infancy and age 30, any increase due to air pollution would make a negligible contribution to the total population LE. Infant mortality is a different matter and it is treated separately, requiring different methods for the epidemiological studies and for the analysis.

The above calculation is directly applicable to steady state exposures and Eq.(6.9) yields the LE loss due to constant exposure during an entire lifetime. The results of Pope *et al.* also correspond to steady state exposure because time variation of the concentration data was not considered. However, it is also of interest to consider time-varying exposures, especially for applications to environmental policy. Furthermore, the real exposure of the cohorts studied by Pope *et al.* declined during the study period and it is not obvious how this may have affected the results.

For these reasons, Leksell and Rabl (2001) extended the above framework by accounting for time-varying concentrations and impacts, in order to derive a correction factor for the exposure changes of Pope *et al*. (1995). They did this by introducing the time constant(s) for the decrease of risk after exposure, based on estimates from studies of smoking (such decrease of risk occurs if the body repairs some of the damage). Even though the uncertainty of time constant(s) inferred from smoking studies is large, it turns out to have almost no effect on the resulting LE loss, as Leksell and Rabl confirmed by a sensitivity analysis.

Leksell and Rabl showed that the LE loss is almost exactly proportional to exposure, defined as time integral of the concentration weighted by an exponential decay factor; the time distribution of the concentration itself does not matter. For comparison purposes it is therefore convenient to state the LE loss for exposure during one year to a concentration increment of 1 $\mu g/m^3$ of PM_{10}; the result can readily be scaled to other exposures. For adults, a good approximation can be obtained using the steady state analysis if one divides the loss due to a constant life time exposure by the life expectancy at birth.

Leksell and Rabl also evaluated how the result could change for different population data. As shown by the examples in Table 6.3, variations due to differences in life table data have only a relatively small effect. This table also shows the mortality data for the respective populations, as expressed by the parameters α and β, fitted by Leksell and Rabl to the life table data for the age specific mortality $\mu(x)$ at age x according to the Gompertz model which is remarkably accurate for all populations above age 30

$$\mu(x) = \exp(\alpha + \beta\, x). \qquad (6.10)$$

Despite a more than twofold difference between male and female mortality in the USA, the LE loss is much the same, 2.56E-04 YOLL for females vs 2.73E-04 YOLL for males, per person per $\mu g/m^3$ of PM_{10} per yr. The largest differences in Table 6.3 are between China (2.04E-04 YOLL) and Russia (3.59E-04 YOLL). Note that differences in LE loss are caused not only by differences in mortality but also by differences in the age distribution.

After a recent recalculation of the LE loss implied by the relative risk of Eq. (6.5), we take the SCR slope for chronic mortality as

$$s_{CR} = 4.0\text{E-4 YOLL}/(\text{pers}\cdot\text{yr}\cdot\mu g/m^3) \text{ for } PM_{10}\,. \qquad (6.11)$$

The same result has been obtained with an independent calculation by Miller and Hurley (2003), using slightly different assumptions and a numerical approach.

Table 6.3 Coefficients α and β of the Gompertz model and LE loss from chronic mortality for several populations. LE loss in YOLL/person for an exposure to 1 μg/m^3 of PM_{10} during 1 year as calculated with the real age distribution of each population, assuming that the relative risk of Pope *et al.* (1995) applies equally to all populations. Adapted from Leksell and Rabl (2001) by converting from $PM_{2.5}$ to PM_{10}.

Population	α [per yr^2]	β [per yr]	LE loss YOLL/(person·yr·μg/m^3), PM_{10}
USA, natural causes, male+female [a]	5.38E-05	8.78E-02	2.69E-04 [e]
USA, natural causes, male [a]	7.76E-05	8.59E-02	2.73E-04 [e]
USA, natural causes, female [a]	3.19E-05	9.20E-02	2.56E-04 [e]
EU15, all causes, male + female [a]	3.70E-05	9.24E-02	2.56E-04
Sweden, natural causes, male+female [b]	9.67E-06	1.10E-01	2.25E-04 [e]
France, all causes, male + female [a]	6.66E-05	8.50E-02	2.77E-04
Russia, all causes, male + female [c]	3.96E-04	6.78E-02	3.59E-04
China, all causes, male + female [d]	5.89E-05	9.15E-02	2.04E-04

[a] data for 1995; [b] data for 1993-1997; [c] data for 1996; [d] data for 1998; [e] includes correction factor for natural mortality r_{nat} = 1/0.975

6.3.4 Loss of life expectancy for acute mortality of adults

It is interesting to compare the LE loss from chronic mortality with the one from acute mortality. Even though acute mortality (short-term) studies do not provide any information about the YOLL per death, one can get a rough idea by assuming, for the sake of argument, 6 months per death as population average.

For acute mortality due to PM, the most comprehensive source is HEI (2001) because it lists the ΔRR results of the three most significant studies, as already cited above in section 6.2.3:

- 0.05% per μg/m^3, by Samet *et al*. (2000), based on the 90 largest US cities;
- 0.06% per μg/m^3, by Katsouyanni *et al*. (1997), based on APHEA results for 12 European cities; and
- 0.07% per μg/m^3 by Levy *et al*. (2000), based on a meta-analysis of 29 studies in 23 locations in Europe and North and South America;

all three referring to PM_{10}. Here we take the central value of

$$\Delta RR = 0.06\% \text{ per } \mu g/m^3 \text{ of } PM_{10} \quad (6.12)$$

based on the WHO meta-analysis of European studies (Anderson *et al*., 2004), which recommends a CRF very close to the original estimates. However, we do not add the result to those of chronic mortality because it is already included in the latter by virtue of the design of the chronic mortality studies.

Taking 10000 deaths/yr per million as a typical value of the reference mortality rate, this implies an average loss of

10000 deaths/yr per million • 0.0006 per $\mu g/m^3$ • 0.5 YOLL/death
= 3E-6 YOLL per person per $\mu g/m^3$ of PM_{10} per yr, if 6 months per death.

This is only about 1% of the total found by long-term studies. With any reasonable assumption for the YOLL per acute death, one finds that the mortality observed by short-term studies is at most a small contribution to the total impact (and in any case it is included in the results of the long-term studies by their very design). The smallness of acute mortality is entirely plausible when one considers what time series studies would be able to observe about mortality from smoking if applied in a hypothetical country where cigarette sales were forbidden on Sundays.

For O_3 only acute mortality has been measured so far with sufficiently firm results. The WHO meta-analysis (Anderson *et al*., 2004) provided the following CRF of an increase in all-cause mortality

0.3% (95% CI 0.1-0.43%) per 10 $\mu g/m^3$ increase in the daily maximum 8-hour mean O_3.

This is used by ExternE, assuming linearity without threshold.

6.3.5 Infant mortality

Woodruff *et al*. (1997), a US cohort study of 4 million infants, showed that post neonatal infant mortality, between the ages of one month and one year, was associated with mean outdoor concentrations of PM_{10} in the first two months of life, giving a CRF for change in all-cause infant mortality of

4% per 10 $\mu g/m^3$ PM_{10} (95% CI 2% - 7%)

6.4 Morbidity

6.4.1 Morbidity due to PM

Morbidity – general methodological remarks

The general approach to estimating the effects of PM (or ozone) on morbidity uses the relative risk found in the epidemiological studies, expressed as % change in end-point per (10)µg/m^3 PM_{10} (or $PM_{2.5}$) and links this with (i) the background rates of the health end-point in the target population, expressed as new cases (or events) per year per unit population – say, per 100,000 people; (ii) the population size and (iii) the relevant pollution increment, expressed in µg/m^3 PM. Results are then expressed as estimated new or 'extra' cases, events or days per year attributed to PM.

Combining the relative risk with the background rates to give a single CRF expressed as:

> number of (new) cases, events or days per unit population (say, per 100,000 people) per (10) µg/m^3 annual average PM_{10} (or $PM_{2.5}$) per annum.

For many health end-points, reliable data on background rates of morbidity in the EU-25 target population are not readily available. One strategy then is to use other general epidemiological studies of that health end-point – not necessarily studies of air pollution and health – to provide estimates of background rates, for example the International Study of Asthma and Allergies in Children (ISAAC) and, for adults, the European Community Respiratory Health Study (ECRHS).

Another approach is to estimate a CRF from the location where the relevant epidemiological studies were carried out and then transfer and use that CRF for quantification in the wider European target population. The two approaches have been used (for different health end-points) in the CAFE-NEEDS methodology. Otherwise, few if any morbidity end-points would have been quantifiable.

New cases of chronic bronchitis and long-term exposure to PM

The US Seventh Day Adventist Study (AHSMOG: Adventist Health Smog) study examined people on two occasions about ten years apart, in 1977 and again in 1987/88. Chronic bronchitis was defined as reporting chronic cough or sputum on most days, for at least three months of the year, for at least two years. New cases of chronic bronchitis were defined as those which met the criteria in 1987/88 but not in 1977. Using a RR from Abbey at al. (1995a, Table 6) and a background incidence rate (adjusted for remission of chronic bronchitis symptoms) of 0.378% estimated from Abbey *et al*. (1993, 1995a), Hurley *et al*. (2005a) derived an estimated CRF of

New cases of chronic bronchitis per year per 100,000 adults aged 27+
= 26.5 (95%CI -1.9, 54.1) per 10 µg/m^3 PM_{10}

New cases of chronic cardiovascular disease
It is to be expected that ambient PM also affects the development and/or the worsening of chronic cardiovascular disease. However, we have not found suitable studies of long-term exposure to quantify these impacts, other than those impacts which result in earlier mortality.

Respiratory hospital admissions (RHAs: ICD 460-519)
Hurley *et al*. (2005a) used all-ages data, both for RR and for background rates, derived from APHEIS-3 (Medina *et al.,* 2005), based on eight European cities. Together they imply a CRF:

Annual rate of attributable emergency RHAs
= 7.03 (95% CI 3.83, 10.30) per 10 µg/m^3 PM_{10} per 100,000 people (all ages)
6.8 Cardiac hospital admissions (ICD 390-429)

CAFE-NEEDS quantified an effect of PM_{10} on cardiac admissions, using a RR based on APHEA-2 results from eight cities in Western and Northern Europe (Le Tetre *et al*., 2002) and a Europe-wide annual rate of emergency cardiac admissions estimated as the arithmetic mean of rates from eight European cities derived from the Appendices of the APHEIS-3 report (Medina *et al*., 2005). Together these imply a CRF:

Annual rate of attributable emergency cardiac hospital admissions
= 4.34 (95% CI 2.17, 6.51) per 10 µg/m^3 PM_{10} per 100,000 people (all ages)

Emergency room visits
Hurley *et al*. (2005a) did not attempt to quantify a relationship between emergency room visits and PM.

Consultations with primary care physicians (general practitioners)
Studies in London have linked daily variations in ambient PM with consultations with primary care physicians for asthma (but not for lower respiratory diseases) (Hajat *et al*., 1999) and for upper respiratory diseases, excluding allergic rhinitis (Hajat *et al*., 2002). These studies were based on numbers of people consulting (including home visits) in a 3-year period 1992-94, among about 282 000 registered patients from 45-47 general practices in the Greater London Area.

Because of differences in health care systems, it is difficult to know to what extent these relationships are transferable within Europe. Hurley *et al*. (2005a) therefore proposed that they be used only in sensitivity analyses, to help assess if these end-points are important.

Consultations for asthma

Separately by age-group, (i) RRs for warm season, adjusted for other factors (Hajat *et al*., 1999), (ii) mean daily numbers of consultations for asthma in the warm season and (iii) numbers of registered patients, were linked and the results expressed as annual CRFs to give

1.18 consultations (95% CI 0, 2.45) for asthma, per 1000 children aged 0-14
0.51 consultations (95% CI 0.2, 0.82) for asthma, per 1000 adults aged 15-64
0.95 consultations (95% CI 0.32, 1.69) for asthma, per 1000 adults aged 65+
per 10 $\mu g/m^3$ PM_{10}, per year.

Consultations for upper respiratory diseases (URD), excluding allergic rhinitis (ICD 460-3; 465; 470-5 and 478)

Analyses by Hajat *et al*., (2002), adjusted for season, day-of-the-week effects and climate, showed statistically significant associations between PM_{10} and consultations by adults and by elderly people. Estimates for children, not statistically significant but quite close to it, are included for completeness. These results, and background rates, were used to derive the following CRFs for attributable consultations for URD (excluding allergic rhinitis)

4.0 consultations (95% CI -0.6, 8.0) per 1000 children aged 0-14
3.2 consultations (95% CI 1.6, 5.0) per 1000 adults aged 15-64
4.7 consultations (95% CI 2.4, 7.1) per 1000 adults aged 65+
per 10 $\mu g/m^3$ PM_{10}, per year:

Restricted activity days and associated health end-points

Ostro (1987) and Ostro and Rothschild (1989) used data on adults aged 18-64 from six consecutive years (1976-81) of the US Health Interview Study (HIS), a multi-stage probability sample of 50,000 households from metropolitan areas of all sizes and regions throughout the USA (Ostro and Rothschild, 1989). Within the HIS, RADs are classified according to severity as (i) bed disability days; (ii) work or school loss days and (iii) minor restricted activity days (MRADs), which do not involve work loss or bed disability but do include some noticeable limitation on 'normal' activity.

Restricted activity days (RADs)

Ostro (1987) studied both RADs and work loss days (WLDs) among adults aged 18-64 in separate analyses for each of the six years 1976-81. A weighted mean coefficient for RADs was linked to estimated background rates of, on average, 19 RADs per person per year (ORNL/RFF, 1994) to give an estimated CRF of:

Change of 902 RADs (95% CI 792, 1013) per 10 $\mu g/m^3$ $PM_{2.5}$
per 1,000 adults at age 15-64.

In the main analyses of CAFE CBA, this CRF was applied to people at ages 15-64, as in the original study. In sensitivity analyses, the same CRF was used but applied to all ages, on the grounds that it is unlikely that health-related restrictions on activity cease at age 65.

Minor restricted activity days (MRADs) and work loss days (WLDs)
As an alternative, Hurley *et al*. (2005) also derived CRFs for work loss days (WLDs) from Ostro (1987) and minor RADs from Ostro and Rothschild (1989) to give, respectively,

Change of 207 WLDs (95%CI 176-238) per 10$\mu g/m^3$ $PM_{2.5}$ per year
per 1000 people aged 15-64 in the general population
and
Change of 577 MRADs (95% CI 468-686) per 10$\mu g/m^3$ $PM_{2.5}$ per year
per 1000 adults aged 18-64.

Medication (bronchodilator) usage by people with asthma
WHO (2004) concluded that there is sufficient evidence to assume a causal relationship between air pollution exposure and aggravation of asthma in children. On that basis, the CAFE-NEEDS quantification of Hurley *et al*. (2005a) proposes CRFs for increased medication usage in people with asthma, although the specific evidence is weak. Separate results were given for children and for adults.

Effects in children aged 5-14 years
Hurley *et al*. (2005) linked an estimated RR from the WHO meta-analysis (Anderson *et al*., 2004), which was dominated by the PEACE study and not statistically significant, with estimates of the mean daily prevalence of bronchodilator usage among panels of school-children who meet the PEACE study criteria, to give a CRF of:

Annual change in days of bronchodilator usage
= 180 (95% CI -690, 1060) per 10 $\mu g/m^3$ PM_{10}
per 1000 children aged 5-14 years meeting the PEACE study criteria.

European data from the International Study of Asthma and Allergies in Childhood (ISAAC Steering Committee, 1998) were used to estimate that approximately 15% of children in Northern and Eastern Europe, 25% in Western Europe, met the PEACE study inclusion criteria.

Effects in adults aged 20+
A RR from the WHO meta-analysis (Anderson *et al*., 2004) was linked with estimates of (i) the mean daily prevalence of bronchodilator use by people with asthma and (ii) the percentage of adults with asthma of a severity comparable to that of the Dutch

panels on whom the RR was based, to give an estimated CRF for change in bronchodilator usage days:

912 (95% CI -912, 2774) per year per 10 $\mu g/m^3$ PM_{10}
per 1000 adults aged 20+ with well-established asthma (say, 4.5% of the adult population).

Lower respiratory symptoms (LRS), including cough, in adults with chronic respiratory disease
A random effects meta-analysis of results from five panels was linked to both (i) estimates of the mean daily prevalences of LRS, including cough, in symptomatic panels, based on the studies underlying the RR, and (ii) estimates of the percentage of people qualifying for such panels, using data from ECRHS (1996), to give an estimated CRF:

Annual increase of 1.30 (95% CI 0.15, 2.43) symptom days (LRS, including cough) per 10 $\mu g/m^3$ PM_{10}
per adult with chronic respiratory symptoms (approx 30% of the adult population).

Lower respiratory symptoms (LRS), including cough, in children in the general population
The recent systematic review by Ward and Ayres (2004) very strongly suggests that effects of PM on respiratory symptoms should be quantified for children generally and should not be confined to children with chronic symptoms. Hurley *et al.* (2005a) combined RRs from Ward and Ayres (2004) with an estimate of the mean daily prevalence of LRS, including cough, based on two general population Dutch studies of children (van der Zee *et al*., 2000; Hoek and Brunekreef, 1995), to give an estimated CRF:

Change of 1.86 (95% CI 0.92, 2.77) extra symptoms days per year
per child aged 5-14, per 10 $\mu g/m^3$ PM_{10}.

Acute respiratory symptoms in the population generally
Hurley *et al*. (2005a) quantified acute respiratory symptoms in adults with chronic respiratory disease rather than in adults generally. However, for sensitivity analyses only, CAFE-NEEDS also included some estimates of the effect of PM on symptom days in the general population, based on Krupnick *et al*. (1990), which had previously been used, e.g. in European Commission (1995), to give

Annual change in symptom days per 1000 people at risk (all ages)
= 4650 (95% CI 210, 9090) per 10 $\mu g/m^3$ PM_{10} .

It is likely that this is a high estimate of the effects of PM on respiratory symptoms, especially for application in Europe. It was included in CAFE CBA with the intention that it be used only for sensitivity analyses, to indicate how big the effect might be.

6.4.2 Morbidity due to O_3

Effects on morbidity of long-term exposure to ambient ozone

There is no strong or quantifiable evidence that long-term exposure to ozone is associated with health effects additional to those which are the aggregate over time of the effects of short-term exposure, i.e. of daily variations in ozone. Consequently, no CRFs linking long-term exposure to ozone and health were proposed by Hurley *et al*. (2005a).

Framework issue: ozone pollution metric used

The WHO evaluations (WHO, 2003, 2004) concluded that there was no evidence for a threshold in the relationship between daily variations in ozone and mortality. However, these evaluations also recognised that, at lower concentrations of daily ozone, there was little evidence on which to base any judgment. Consequently, the TFH of WHO-UNECE decided that, in the core analyses, the effects of daily ozone on mortality should be quantified only at ozone concentrations higher than 35 ppb (70 $\mu g/m^3$), considered as a daily maximum 8-hour mean ozone concentration. In practice, this means that effects are quantified only on days when the daily ozone concentration (maximum 8-hour mean) exceeded 70 $\mu g/m^3$, and then only the increment exceeding 70 $\mu g/m^3$ is used for quantification. This increment, aggregated over all days of the year, was called SOMO35 and is the exposure metric used for quantification in CAFE-NEEDS.

WHO-UNECE emphasised that the use of a cut-off should not be interpreted as acceptance of a threshold and recommended also that, for sensitivity analyses, effects be estimated with a cut-off of zero. In CAFE-NEEDS these recommendations regarding no threshold but with a cut-off for daily ozone, originally developed in the context of daily mortality, were subsequently applied to all CRFs.

ExternE assumes linearity without threshold for all the CRFs for ozone.

Mortality at all ages from short-term exposure to O_3

The WHO meta-analysis (Anderson *et al*., 2004) provided a RR of an increase in all-cause mortality of

0.3% (95% CI 0.1-0.43%)
per 10 $\mu g/m^3$ increase in the daily maximum 8-hour mean O_3.

This RR, which applies to all ages, was used in CAFE CBA, in line with guidance from TFH of WHO-UNECE.

Respiratory hospital admissions (RHAs)
Anderson *et al*. (2004) used results from five cities in Western Europe to estimate the change in all RHAs in various age groups in relation to daily variations in O_3 (8-hr daily average) with an effect – close to statistical significance – for elderly people only, giving a RR of 0.5% (95% CI -0.2%, 1.2%) per 10 μg/m^3 O_3 (8-hr daily max) in people aged 65+. Background rates in people aged 65+ were taken from the APHEIS second year report (APHEIS, 2002), giving a CRF:

Annual rate of attributable emergency RHAs per 100,000 people at age 65+
= 12.5 (95% CI -5.0, 30.0) per 10 μg/m^3 O_3 (8-hr daily average).

Cardiovascular hospital admissions
There is no strong or quantifiable evidence that daily variations in ozone are associated with cardiovascular hospital admissions or, indeed, with other cardiovascular morbidity end-points.

Emergency room visits
Hurley *et al*. (2005a) did not attempt to quantify a relationship between emergency room visits and ozone.

Consultations for allergic rhinitis (ICD9 477), with primary care physicians (general practitioners)
Hajat *et al*. (2001) studied consultations for allergic rhinitis (ICD9 477) and found that relationships with ozone (8-hr daily max) were strongest using a cumulative index incorporating O_3 over four consecutive days, with lags 0-3 days, based on numbers of people consulting (including home visits) in a 3-year period 1992-94, among about 282,000 registered patients from 45-47 general practices in the Greater London Area. Hurley *et al*. (2005) used these results, applying them as if to a single day's pollution, and linked them to mean daily numbers of consultations and numbers of registered patients to give estimates of change in annual consultations for allergic rhinitis of:

3.03 consultations (95% CI 1.89, 4.29) per 1000 children aged 0-14
1.60 consultations (95% CI 1.22, 2.03) per 1000 adults aged 15-64
per 10 μg/m^3 O_3 .

Because of differences in health care systems, it is difficult to know to what extent these relationships are transferable within Europe. We recommend that they be used in sensitivity analyses only, to help assess if these end-points are important.

Minor restricted activity days (MRADs)
For current urban workers aged 18-64, Ostro and Rothschild (1989) reported relationships between minor restricted activity days (MRADs) and ozone (two-week averages of the daily 1-hr max, in µg/m^3). The weighted mean coefficient for ozone, adjusted for $PM_{2.5}$, from separate analyses of each of the six years 1976-81 was linked with a mean background rate of 7.8 MRADs per year among people in employment aged 18-64 (Ostro and Rothschild, 1989) to give an estimated CRF:

Increase in MRADs = 115 (95% CI 44, 186)
per 10 µg/m^3 ozone (8-hr daily average) per 1000 adults aged 18-64 per year.

Issues of uncertainty are addressed, as for other end-points, in CAFE CBA Vol. 3 (Holland *et al*., 2005a); in UNICE's letter of concerns about the CAFE Methodology, and in the CAFE CBA team's response (Hurley *et al*., 2005b).

Medication (bronchodilator) usage by people with asthma
As for PM, the CAFE-NEEDS quantification of Hurley *et al*. (2005a) proposes CRFs for increased medication usage in people with asthma, although the specific evidence is weak. Separate results were given for children and for adults.

<u>Effects in children aged 5-14 years</u>
A RR was derived from Just *et al*. (2002), a small study of 82 children with medically diagnosed asthma in Paris in early summer 1996, and the only European study giving a relationship between daily ozone (8-hr daily mean) and medication use in children with asthma. Background rates were derived from Gielen *et al*. (1997) and from Just *et al*. (2002), with different functions reflecting higher prevalences of childhood asthma in Western Europe than in Northern and Eastern Europe (ISAAC, 1998). These results were combined to give an estimated CRF of:

Annual change in days of bronchodilator usage
124 (95% CI 18, 227) in Northern and Eastern Europe;
310 (95% CI 44, 569) in Western Europe.
per 10 µg/m^3 O_3 per 1000 children age 5-14 years (general population):

Two points should be noted. First, while the effects occur only in children with asthma, the CRF was derived to apply to the general population. Secondly, as noted, Just *et al.* (2002) is a small study in one location. Furthermore, the estimated odds ratio is very high, compared with other end-points. The study may well be unrepresentative; it may be best to consider it as an upper limit, i.e. for sensitivity analysis only.

Effects in adults aged 20+ with asthma

Hiltermann *et al*. (1998) gave results linking daily max 8-hr moving average O_3 with daily prevalence of bronchodilator usage. This was positive but not statistically significant (OR 1.009 per 10 µg/m^3 O_3; 95% CI 0.997, 1.020) at the selected lag of 1 day, although when 7-day cumulative ozone was considered, the estimated effect was higher and statistically significant. Background rates were estimated using results from Hiltermann *et al*. (1998) and from the European Community Respiratory Health Survey (ECRHS, 1996). These data were linked to give an estimated CRF:

> Change in days of bronchodilator use of
> 730 (95% CI -255, 1570) per 10 µg/m^3 O_3
> per 1000 adults aged 20+ with well-established asthma (approximately 4.5% of the adult population).

Acute respiratory symptoms in children in the general population

Work in progress by the Committee on the Medical Effects of Air Pollutants (COMEAP) in the UK suggests that there is convincing evidence that daily variations in ozone are associated with lower respiratory symptoms (LRS), including cough, and that these effects are not restricted to people with chronic respiratory symptoms such as asthma (Heather Walton, 2004, personal communication). The CAFE-NEEDS Methodology Report (Hurley *et al*., 2005a) used a small general population study of 91 children in Armentieres, Northern France (Declerq and Macquet, 2000), to quantify relationships linking (i) daily prevalence of cough and phlegm and (ii) lower respiratory symptoms (LRS), excluding cough, with 8-hr daily max O_3. The relevant RRs were linked with background rates derived from Hoek and Brunekreef (1995) to give CRFs:

> change of 0.93 (95% CI -0.19, 2.22) cough days
> and 0.16 (95% CI -0.43, 0.81) days of LRS (excluding cough)
> per child aged 5-14 years (general population), per 10 µg/m^3 O_3, per year.

6.5 Toxic Metals, Dioxins and Other Pollutants

6.5.1 General remarks

The most toxic metals are As, Cd, Cr (in oxidation state 6, designated as CrVI), Hg, Ni and Pb. They have a variety of adverse health impacts but, at the present time, the only end-points that can be quantified are cancers for As, Cd, CrVI and Ni, and neurotoxic impacts for Pb. The major impacts of Hg are also neurotoxic but their quantification still poses too many problems at the present time. Among the impacts of dioxins are endocrine disruption and cancers but only the latter can be quantified at the present time. We also consider cancers due to inhalation of benzene, formaldehyde, butadiene and benzo(a)pyrene.

Data can be found at the IRIS web site of the US EPA (http://www.epa.gov/iriswebp/iris/index.html). The CRFs for cancers due to inhalation given by EPA are stated as unit risk factors (URF), defined as the probability, per $\mu g/m^3$ of ambient concentration, of getting a cancer due to a lifetime exposure (taken as 70 yr). With our definition of the CRF as impact for a 1 yr exposure, the slope s_{CR} is the unit risk divided by 70.

The scientific evidence usually consists of animal studies and some epidemiological studies of workers exposed to high concentrations. There are major methodological issues when using either occupational and/or animal studies for quantitative human risk assessment; see, for example, US EPA (1996) or HEI (1995). The overview by Nauss *et al*. in HEI (1995) is particularly useful. Issues to be considered include that:

- the reliability of risk estimates in occupational studies depends crucially on the reliability of estimated long-term exposures of the study subjects, and this can vary considerably;
- use for public health risk estimation requires extrapolation both to low concentrations and to possibly more susceptible individuals;
- quantitative use of risk estimates from animal studies may also involve low-dose extrapolation and quantitative animal-to-human scaling.

It follows that unit risk estimates depend not only on the availability and reliability of human and animal evidence on the carcinogenicity of the pollutant, but also on the models and assumptions used in extrapolating from that evidence and, more generally, on the judgment of the group of experts making the classification. These difficulties have led to substantial diversity in the acceptance of quantified risk estimates for development of cancer.

Among the URFs proposed by various expert groups, those proposed by the US EPA are commonly quoted. These are used here as well for all carcinogens under consideration, to ensure consistency. In extrapolating from high to low exposures, the EPA has traditionally used the linearised multistage model, which assumes no threshold, and (as its name suggests) a linear dose-response relationship in the low-dose region. It may therefore over-estimate impacts at low exposures. A proposal how to overcome this conservatism is made by Crettaz et al. (2002) and recently applied by Bachmann (2006) in the realm of external cost assessments. In the following, however, this deviation from the US EPA approach has not been considered.

For many of these pollutants, in particular dioxins and the most toxic metals (As, Cd, CrVI, Hg, Ni and Pb), the dose from ingestion of food is about two orders of magnitude larger than the inhalation dose. However, the health impact per dose can be different depending on the intake mode: for example, according to current knowledge Cd, CrVI and Ni are carcinogenic only via inhalation.

For CRFs determined by epidemiological studies, the question arises whether the effect of the ingestion dose should be added to that of inhalation. This depends on what exactly was measured in the epidemiological study. Typically the study population was exposed simultaneously via inhalation and ingestion. Even if the result of a study is stated as CRF, i.e. in terms of ambient air concentration, it may in fact reflect the total dose. If the ratio of inhalation and ingestion for the general population is different from that of the study population, one does not know how to apply the CRF unless one can make reasonable assumptions about the separate inhalation and ingestion doses of the study population and the relative effectiveness of these two dose routes.

6.5.2 As, Cd, Cr and Ni

For the carcinogenic metals, As, Cd, Cr (in oxidation state 6) and Ni, the unit risk factor (URF) is shown here in the third line of Table 6.4 and the CRF slope s_{CR} in the fourth. At the present time the evidence for cancers due to ingestion of Cd, Cr and Ni is not sufficiently convincing for EPA to indicate a DRF.

Table 6.4 CRFs and DRFs, per kg emitted, for the carcinogenic metals. Unit risk and slope factor from IRIS http://www.epa.gov/iris.

	As	Cd	Cr-VI [a]	Ni
Inhalation				
URF [cancers/(pers·70yr·μg/m^3)]	4.30E-03	1.80E-03	1.20E-02	2.40E-04
s_{CR} [cancers/(pers·yr·kg/m^3)]	6.14E+04	2.57E+04	1.71E+05	3.43E+03
Ingestion				
slope factor [cancers/(mg/(kg$_{body}$·day))]	1.50E+00			
s_{DR} [cancers/kg]	1.07E+00 [b]			

[a] if only total Cr emission is known, one must estimate the fraction in the VI oxidation state; typical numbers are 11% for coal-fired and 18% for oil-fired power plants, according to EPA (www.epa.gov/ttncaaa1/t3/meta/m28497.html, www.epa.gov/ttncaaa1/t3/meta/ m27812. html).
[b] for inorganic As

Ingestion of As is considered carcinogenic with slope factor 1.5 per mg/(kg.day). Since the slope factor indicates the lifetime risk due to ingesting the same dose every day for 70 yr, we need to divide by 70×365 days and the average weight of 55 kg/pers (adults and children) to obtain the DRF in our units. At the present time EPA and the International Agency for Research on Cancer do not provide any information on the carcinogenicity of organic As. Most of the ingestion dose is organic, with the exception of drinking water which is inorganic.

6.5.3 Pb

The most important health impact of Pb seems to be IQ decrement. The dose-response function is quite well-determined, thanks to a meta-analysis by Schwartz (1994) who found a decrement of 0.026 IQ points for a 1 μg/L increase of Pb in blood, a relation that appears to be linear without threshold. More recently a study designed to identify effects at the lowest doses found an even larger effect, 0.055 IQ points per 1 μg/L, without any threshold (Lanphear *et al*., 2000). Here we continue to use 0.026 IQ points per 1 μg/L, being based on a meta-analysis rather than a single study.

To relate blood level to exposure and dose we have found two options and so we present two calculations. The first is a relation recommended by a recent UK review (EPAQS, 1998) which finds that a 1.0 $\mu g/m^3$ incremental exposure to Pb in ambient air increases the blood level by 50 μg/L, not very different from values in an earlier review by Brunekreef (1984). Combined with 0.026 IQ points per 1 μg/L, this implies a loss of 1.3 IQ points per child per $\mu g/m^3$.

We also need to consider the time window during which an exposure causes damage. The sensitivity of the brain to Pb is greatest during the first two years of life, although the precise time distribution of the damage is not known. However, this does not matter since the result of Schwartz expresses the total impact in a population due to a constant exposure. Furthermore, the half life of Pb in blood and other soft tissues is relatively short, about 28-36 days (although much longer in bones) (WHO, 1995). Thus, for the purpose of damage calculations, one can equally well assume that the damage is incurred during a one year exposure by infants between the ages of zero and one only, or during a three year exposure between the ages of zero and three. To see that the effect is the same, note that the percentage of the population between zero and three is essentially three times the percentage between zero and one, the latter being 1.1% in the EU. If the sensitive period is only one year, the loss due to a one year exposure is 1.3 IQ points/($\mu g/m^3$) × 1.1% of population of EU. If the sensitive period is three years, the affected cohort is essentially three times as large but the damage rate three times smaller, so the loss due to a one year exposure is (1.3 IQ points/($\mu g/m^3$))/3 × (3 × 1.1% of population of EU), essentially the same. To express the CRF slope in a form consistent with this paper, i.e. relative to the entire population, we therefore multiply the 1.3 IQ points/($\mu g/m^3$) by the fraction of the population that is affected (1.1% per year), to obtain

$$s_{CR} = 1.43\text{E-2 IQ points/(pers}\cdot\text{yr}\cdot(\mu g/m^3)) \quad \text{(includes ingestion)}. \qquad (6.13)$$

We use this function without adding a further contribution from ingestion because the above relation between ambient concentration and blood level has been observed in populations who also received a dose from ingestion; thus the ingestion dose is implicitly taken into account.

The second option is a relation between blood level Pb and ingestion dose, published by WHO (1995). Surprisingly the blood level per ingested quantity is higher at low doses, perhaps because of increased excretion at higher dose or storage in bones. Here we use the level found at the lower dose, 72 μg/L for infants who ingest 17 μg/day, or 4.2 μg/L per ingested μg/day. Together with the above mentioned 0.026 IQ points per 1 μg/L increase of blood Pb, this implies a loss of 0.026 IQ points × 4.2 (μg/L)/(μg/day) × (1 yr/365 days) = 3.02E-04 IQpoints/(μg/yr) per child. As in the argument leading to Eq.(6.13) we multiply this number by 1.1%, the fraction of the total population below 1 yr of age and sensitive to Pb, to obtain a DRF slope of

$$s_{DR} = 3.30\text{E}{+}03 \text{ IQpoints}/(\text{kg}_{\text{absorbed}}). \quad (6.14)$$

Again the duration of the sensitive period during infancy does not matter. We have more confidence in the relation of blood level with ingestion than with inhalation. The relation with inhalation appears less reliable, as the inhalation/ingestion ratio is likely to be quite variable with site, and over time as well, especially with the phasing out of leaded gasoline.

6.5.4 Dioxins

Dioxin is one of the most thoroughly studied of all of the pollutants. Several human epidemiological studies and numerous studies in experimental animals have been carried out. There can be acute as well as chronic effects. Dioxins cause changes in laboratory animals that may be associated with developmental and hormonal effects; however, the mechanism of carcinogenicity is unclear. Whether the biochemical changes may result in adverse health effects in people and at what concentrations is not very well known.

In laboratory experiments with animals, TCDD (tetrachlorodibenzo-p-dioxin) has been found to be one of the most potent toxins known, with LD_{50} ranging from 0.6 to 3000 μg per kg of body weight for different mammals (LD_{50} is the dose that kills half of a test group) (Tschirley, 1986). This wide range of values suggests that extrapolation from one animal species to another is quite uncertain.

However, Tschirley (1986) cites other pieces of evidence that are directly relevant to humans. In particular, there is an experiment on prisoners, performed in days past when such experiments were not yet considered immoral. In one such experiment 60 prisoners were exposed, twice within 2 weeks, to a TCDD dose of 3 to 114 ng per kg_{body}. No symptoms were observed. An interesting additional data point, in the same reference, comes from another experiment with 10 volunteer prisoners who were exposed to a much larger dose of 107 μg per kg_{body}. Eight of them developed chloracne but no other symptoms were noted.

These numbers indicate that humans are certainly not among the most sensitive species as far as acute dioxin toxicity is concerned. They also suggest that the threshold for non-cancer toxicity is at least 114 ng/kg_{body}/(70 × 365 days) = 8.92E-06 $\mu g/kg_{body}\cdot day$, if one spreads the acute dose in the above experiment over a lifetime of 70 years. That should be a lower bound of the threshold because toxicity tends to be reduced if a dose is administered at a lower rate (e.g. fifty sleeping pills can kill if taken all at once). This number is close to the threshold 10 $pg/kg_{body}\cdot day$ specified by the WHO (1987) as tolerable daily intake.

As for cancers, dioxins (2,3,7,8-TCDD and HxCDD) were said by EPA to be "the most potent carcinogen(s) evaluated by the EPA's Carcinogen Assessment Group". The slope factor is 1.0E+06 cancers/(mg/($kg_{body}\cdot day$)) (EPA 2000).

6.5.5 Benzene, butadiene, benzo(a)pyrene and formaldehyde

Benzene

Benzene is classified by IARC as Category 1, a known human carcinogen. However, risk quantification is complicated by lack of 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. Individual variation in susceptibility or metabolism may therefore influence the risk at any given exposure.

There are many occupational studies investigating exposure to benzene and development of cancer, especially leukaemia. The US EPA risk assessment for benzene gave a unit risk factor of 8×10^{-6} cancers/(pers·70yr·$\mu g/m^3$) (US EPA, 1990). Many different risk estimates have been derived, using different assumptions about the pattern of exposures, the shape of the CRF, and so the extrapolation to low concentrations. These are similar to the estimates of Crump (1994) who gives a range of 4.4 to 7.5×10^{-6} cancers/(pers·70yr·$\mu g/m^3$) for the URF of leukaemia. There is no convincing evidence of chronic non-cancer effects at ambient concentrations.

1,3-Butadiene

1,3-butadiene is potentially carcinogenic to both the white and red cell systems. Animal studies have shown that it is carcinogenic in mice and other rodents. There is however wide discrepancy in metabolism between different species, complicating extrapolation to humans. Although the available animal evidence for 1,3-butadiene and comparison with substances of similar chemical structure would support the classification of butadiene as a human carcinogen, the available human data is limited; and 1,3-

butadiene is classified by IARC as Category 2a, probable human carcinogen (IARC Monographs Volume 54, 1992). Irritant effects also occur but only at concentrations much higher than those relevant to ExternE.

1,3-butadiene is a major ingredient of synthetic rubber and, being volatile, the route of absorption is primarily inhalation. The epidemiological evidence consists mostly of mortality studies which use qualitative estimates or exposure categories rather than estimates of actual lifetime exposures, and with limited consideration of other workplace exposures. There is no evidence available on cancer risks to the general population from ambient exposures. The human studies cannot be used directly in quantified risk assessment because sufficiently reliable estimates of past exposures are not available. The US EPA (1990) URF of 3×10^{-4} cancers/(pers·70yr·$\mu g/m^3$) is based on multi-stage modelling of animal (mice) experimental data. An updated estimate by RIVM (1994) of 0.7 to 1.7×10^{-5} is much lower. However, the contribution of this pollutant to the total damage cost of vehicle emissions is extremely small.

Polycyclic Aromatic Hydrocarbons (PAHs)
These are ring compounds resulting from the incomplete combustion of organic material and which jointly share carbon atoms. They cover a wide range of substances including benzo[a]pyrene (BaP). The relationship between BaP and other PAHs differs for various types of emission but has been shown to be relatively similar in the ambient air of several towns and cities.

There is strong evidence, including from epidemiological studies (e.g. Redmond *et al*., 1972; Hurley *et al*., 1983; Armstrong *et al*., 1994), to suggest that certain components of PAHs, and specifically benzo[a]pyrene, are carcinogenic in humans; and that nitroaromatics as a group pose a hazard to health. In 1986 IARC and the US 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, is labelled as a probable human carcinogen.

As these compounds form complex mixtures and are also absorbed onto particulates, it is difficult to quantify levels of human exposure and so is difficult to estimate risks reliably. Benzo[a]pyrene is the only PAH for which a suitable database is available, allowing quantitative risk assessment. The EPA unit risk factor of lung cancer for BaP is 1×10^{-7} per $\mu g/m^3$ (US EPA, 1990). Limitations in the use of benzo[a]pyrene as an indicator of PAH toxicity in air pollution are that some PAH is bound to particulates, and that some of the gaseous components are not included. WHO (1987) estimated a URF of 8.7×10^{-8} per $\mu g/m^3$; i.e. almost identical to that used by US EPA.

Formaldehyde
Formaldehyde is highly water soluble and most of inhaled formaldehyde is deposited in the lining of the nose. It is a potent irritant and no clear threshold has been defined for

these effects. Available evidence suggests that ambient levels of formaldehyde could produce irritant symptoms to the eyes and respiratory tract in a sub-group of the general population. It is unlikely to cause asthmatic symptoms in healthy subjects at exposures encountered in environmental settings but could potentially exacerbate symptoms. Thus, an occasional mild effect (e.g. symptom day) among sensitive people cannot be ruled out where incremental formaldehyde from transport adds to existing relatively high background levels. However, effects are likely to be small and are difficult to quantify; no CRFs are proposed here.

Formaldehyde is classified as IARC Category 2A, probable human carcinogen (IARC Monographs, Volume 62, 1995). There is however no convincing evidence of an effect at low ambient exposures and possible mechanisms suggest that, in the absence of damage to the respiratory tract tissue, any cancer risks at low ambient concentrations are negligible (WHO, 1997). The US EPA risk assessment for formaldehyde gave a URF of 1×10^{-5} per $\mu g/m^3$ (US EPA, 1990). This URF may substantially over-estimate the true risks from ambient pollution; it is recommended to be used for sensitivity analyses only.

There is limited evidence that formaldehyde may contribute to the development of asthma, especially where there is co-exposure to environmental tobacco smoke (Krzyzanowski *et al*., 1990). However, effects are not well-established and are not reliably quantifiable.

6.6 Health impacts from noise

Noise affects people in a number of ways. There are effects which people perceive, such as the impact on conversation, listening and the enjoyment of outside space. There are effects which are not easily perceived such as the impact of noise on certain aspects of health. Consequences resulting from exposure to transport noise, which affects human life and human health, are quantified by the use of exposure-response functions. A large amount of scientific literature on health and psychosocial effects considering a variety of potential effects of transport noise is available. The scientific basis used here relates to the state of the art summary by De Kluizenaar *et al*. (2001). In their review work, they report risks due to noise exposure in the living environment. Quantitative functions for relative and absolute risks are proposed for the effect categories presented in Table 6.5.

Eight end-points for concrete health effects were identified for stress-related health effects and exposure-response-functions were constructed. The end-points are defined in a way appropriate for economic valuation. They are listed, together with the ER-functions used, in Table 6.6.

Table 6.5 Categorisation of effects and related impact categories.

Category	Measure given	Impacts
Stress-related health effects	RR	Hypertension and ischaemic heart disease
Psychosocial effects	AR	Annoyance
Sleep disturbance	AR	Awakenings and subjective sleep quality

RR = relative risk; AR = absolute risk

Table 6.6 Exposure-response functions for stress-related health effects and sleep disturbance.

End-point	Expected value [a)] (per 1000 adults exposed)
Myocard infarction (MI), fatal, Years of life lost (YOLL)	$0.084 \cdot L_{DEN} - 5.25$
Myocard infarction (non-fatal), days in hospital	$0.504 \cdot L_{DEN} - 31.5$
Myocard infarction (non-fatal), days absent from work	$8.960 \cdot L_{DEN} - 56$
Myocard infarction, expected cases of morbidity	$0.028 \cdot L_{DEN} - 1.75$
Angina pectoris, days in hospital	$0.168 \cdot L_{DEN} - 10.5$
Angina pectoris, days absent from work	$0.684 \cdot L_{DEN} - 42.75$
Angina pectoris, expected no. of morbidity days	$0.240 \cdot L_{DEN} - 15$
Hypertension, days in hospital	$0.063 \cdot L_{DEN} - 4.5$
Sleep disturbance, road traffic	$0.62 \cdot (L_{Aeq,23\text{-}07h} - 43.2)$ [b)]
Sleep disturbance, rail traffic	$0.32 \cdot (L_{Aeq,23\text{-}07h} - 40.0)$ [c)]
Sleep disturbance, air traffic	$0.48 \cdot (L_{Aeq,23\text{-}07h} - 32.6)$ [d)]

Notes: [a)] Threshold is 70 dB(A) L_{DEN} except for [b)] 43.2 dB(A), [c)] 40 dB(A) and [d)] 32.6 dB(A); Other assumptions: MI, 7 years of life lost per fatal heart attack in average; base risk of MI: 0.005; survival probability of MI: 0.7; MI, morbidity: 18 days in hospital per MI, 32 days absent from work; Angina pectoris, base risk: 0.0015; days in hosp.:14 per severe episode; 20 days of morbidity per episode; $L_{Aeq,23\text{-}07h}$ as assessed outside at the most exposed façade.

Sleep disturbance is quantified by calculating the percentage of the exposed population expected to react as highly annoyed by sleep-disturbance. The functions are derived from noise effect surveys on self-reported sleep disturbance and night-time equivalent sound level at the most exposed façade of the dwelling.

Although ER-functions to predict annoyance reactions on the population level are available, they could not be used to date. For the valuation of annoyance impacts, expressed as the share of the population reacting little annoyed, annoyed and highly annoyed, no corresponding monetary value is available where the use of the same definition of annoyance levels was assured. Therefore, another method to value amenity losses due to noise was used based on hedonic pricing.

6.7 Other Impacts

6.7.1 Impacts on building materials

This section draws on Tidblad and Kucera (2001), partly modified by Bert Droste-Franke.

It has been known for several centuries that air pollutants emitted by burning of fossil fuels have a serious impact on buildings. The effects include loss of mechanical strength, leakage and failure of protective coatings due to degradation of materials. Also, the disagreeable appearance in all larger towns of soiled but otherwise beautiful buildings is caused by deposition of particulate matter arising from atmospheric pollution.

For several materials that are frequently used in buildings, dose-response functions have been obtained. A dose-response function links the dose of pollution, measured in ambient concentration and/or deposition, to the rate of material corrosion. They are of outmost importance for development of systems for classification of corrosivity of environments, for mapping of areas with increased risk of corrosion, and for calculation of cost of damage caused by deterioration of materials. In order to be able to calculate costs, a damage function needs to be obtained. A physical damage function links the rate of material corrosion (due to the pollution exposure given by the dose-response function) to the time of replacement or maintenance of the material. Performance requirements determine the point at which replacement or maintenance is considered to become necessary. If the performance requirements can be described in terms of a critical degradation level, it is possible to transform a dose-response function into a damage function. The definitions of dose-response and damage functions mentioned are the official terminology of UN ECE Workshop (1997). In addition, the term exposure-response function has been used within the ExternE project. It refers to a function expressing the maintenance frequency in terms of the critical damage and pollution parameters. It may be regarded as a special damage function where frequency instead of time is the explained variable.

In the following sections the state of the art of impacts of atmospheric pollutants on building materials is briefly summarised. The description is focused on dose-response functions, their transformation into exposure-response functions and interpretation for calculating costs. The text is divided into two sections dealing with degradation and soiling separately and one concluding section describing the combined effects. Soiling is only one of the effects of particulate matter that play an important role in the atmospheric corrosion process. However, these other effects are at the present stage not sufficiently quantifiable.

The dose-response functions presented here from the ICP Materials exposure programme are currently used for the impact pathway assessment. They are primarily developed for the SO_2-dominating situation. Functions which should preferably be used for the multi-pollutant situation, especially if large contributions from particulate matter and/or nitrogen pollutants are expected, are currently derived in the EU 5FP MULTI-ASSESS project. These will be published in the near future. New dose-response functions for soiling have also been developed in the MULTI-ASSESS project.

Degradation of Building Materials

There are many parameters that can influence the damage to materials, which is an interplay between chemical, physical and biological parameters. The present study will focus on the aspects specific to the urban situation, i.e., man-made pollutants and their interplay with natural climatic factors. It is, however, important to recognise that corrosion is a process that occurs even in the absence of pollutants and it is important to quantify to what extent urban conditions affect and accelerate the ”natural” or background corrosion of materials.

Sulphur and nitrogen compounds including secondary pollutants and particulates are the most important pollutants acting as corrosive agents. It has clearly been demonstrated that pollutants enhance the natural corrosion process for several, both metallic and non-metallic, materials. Systematic laboratory exposures in the 1930s demonstrated the corrosive effect of SO_2 on metals. This was later also proved by field exposures and SO_2 was for a long time considered to be the main corrosive pollutant. Today, SO_2 is no longer regarded as the only important corrosion stimulator. Instead, its effect in combination with other gaseous pollutants such as NO_2, O_3, and their reaction products needs to be considered.

Damage Mechanisms

Two types of deposition processes are recognised in atmospheric corrosion, dry deposition and wet deposition, depending on the way in which pollutants are transported from the atmosphere to the corroding surface. Wet deposition refers to precipitation whereas dry deposition refers to the remaining processes including gas-phase deposition and particle deposition. For sheltered conditions, wet deposition and run-off are excluded and the corrosion, i.e., the destruction of the base material, is identical to the formation of the corrosion products. Different climatic parameters influence the deposition rates and are also important in the degradation process.

Climatic Effects

Wind and water can cause erosion and temperature fluctuations result in freezing-thawing cycles. The volume expansion of water turning to ice results in significant stresses to any porous material. Sun radiation can also be a direct factor, especially for the degradation of polymers. Climate has a significant role in degradation of materials, hence the term ”weathering”.

The dry deposition of pollutants is greatly influenced by temperature and relative humidity since they are the main factors that determine the presence of moisture in the absence of rainfall. The time of wetness (TOW) is a commonly used concept for inorganic materials, in particular for metals, that refers to the time when corrosion occurs, i.e., when a moisture layer is present. The degradation process may be considered as discontinuous, only occurring when the surface is sufficiently wet. This concept is useful when describing the degradation process and when classifying different climatic regions from a corrosion point of view but is difficult to calculate from readily available meteorological data. Therefore, recently developed dose-response functions include annual averages of the more easily available parameters temperature and relative humidity.

Dry Deposition Effects

Dry deposition refers to the process by which particles and gases are transferred from the atmosphere to the material surface. The deposition velocity, i.e., the ratio of surface flux to concentration for a particular gas or particle, depends not only on the conditions in the atmosphere but also on the thickness of the moisture layer, the reactivity of the material and the properties of the corrosion products.

Sulphur dioxide is one of the main contributors to the degradation of materials. SO_2 is dissolved in the moisture layer forming sulphite and, after oxidation, sulphate. This process results in an acidification of the moisture layer, which enhances the corrosion process. SO_2 is partly oxidised in the atmosphere and contributes therefore to the wet deposition acidity. Sulphate is also frequently present in corrosion products. The SO_2 deposition rate depends mostly on the material, it is often higher for materials sensitive to degradation, and varies between 0.01 and 2 cm/s.

The role of nitrogen oxides in the degradation of materials has not yet been fully elucidated. Nitrogen is mainly emitted as NO formed in the combustion process and then further oxidised to NO_2 and HNO_3 by photochemical reactions and possibly neutralised by NH_3 forming particulate nitrates (NH_4NO_3). Nitrates are not frequently found in the corrosion products. For unsheltered surfaces this can be explained by the high solubility of many nitrate precipitates. Even if NO_2 in itself is much less detrimental to most materials it can contribute to the acidity of precipitation in a similar way to SO_2.

Ozone (O_3) is a principal pollutant, which in the past was mainly associated with the degradation of natural rubber but most organic materials containing carbon double bonds, such as painted surfaces, polymers and textiles are sensitive to its effect. It is, however, a general oxidant and thus, for inorganic materials O_3 has a synergistic effect in combination with SO_2 as has NO_2.

Particles containing NH_4NO_3, $(NH_4)_2SO_4$ and NH_4HSO_4 play an important role in atmospheric corrosion and this is related to their ability to increase the time of wetness due to their hygroscopic properties. In addition to prolonging the time of wetness, ionic particles enhance the corrosion by providing corrosion stimulators. However, the present dose-response functions do not include the effects of particles as a separable part.

Wet Deposition Effects

The effect of wet deposition can be either detrimental or beneficial, depending on the conditions. The wet deposition has two effects on the corrosion process. On the one hand, it transports chemically active compounds present in rain to the surface, thereby increasing the corrosivity of the moisture layer. On the other hand, it washes away chemically active compounds previously deposited on the surface, with the opposite effect. Thus, for a specific material and environment, choosing a sheltered exposure condition rather than unsheltered may or may not increase the corrosion rate. When the rain acidity is high, the detrimental effect often dominates and the effect is usually quantified in terms of total acid load.

Dose-Response Functions

During the last decades a number of field exposure programmes have been performed. Most of the exposures have been confined to relatively small geographical areas and express the corrosion/degradation on a local or regional basis. Others can be classified as national exposure programmes, where the variation in meteorological parameters may be more extensive. Finally there are a few international exposure programmes, which cover extensive geographical areas. The weak point of most studies is usually the quality of environmental data, which is often inferior to that of the material degradation data. For many materials, numerous dose-response functions have been reported. It is not the intention of the present report to give a full review of all those functions. Instead, recent functions for selected materials obtained within the UN ECE International Co-operative Programme on Effects on Materials, including Historic and Cultural Monuments (ICP Materials) is briefly presented and discussed for the purpose of performing cost calculations. A short description of the programme is also included. For further reference see Tidblad *et al*. (1998). One important task for the programme has been to estimate the relative contribution of dry and wet deposition to the degradation of materials. Therefore, and also because it makes sense from a mechanistic point of view, the dose-response relations are of the form where the corrosion attack, K, is described in terms of dry and wet deposition effects separated as additive terms

$$K = K_{dry} + K_{wet} \qquad (6.15)$$

The dry deposition term is quantified in terms of the parameters SO_2, relative humidity and temperature, whereas the wet deposition is quantified in terms of total amount of precipitation and precipitation acidity.

ICP Materials

The aim of ICP Materials is to perform a quantitative evaluation of the effect of sulphur pollutants in combination with NO_x and other pollutants as well as climatic parameters on the atmospheric corrosion of important materials. This is achieved by measuring gaseous pollutants, precipitation and climate parameters at or nearby each test site and by evaluating the corrosion effects on the materials. A Task Force is organising the programme with Sweden as lead country and the Swedish Corrosion Institute serving as the main research centre. Subcentres in different countries have been appointed, each responsible for their own group of materials. The materials included are:

- Structural metals including steel, weathering steel, zinc, aluminium, copper and bronze.
- Stone materials including Portland limestone and white Mansfield dolomitic sandstone.
- Paint coatings including coil-coated steel with alkyd melamine, steel with alkyd paint, wood with alkyd paint system and wood with primer and acrylate.
- Electric contact materials including nickel, copper, silver and tin as coupons, and Eurocard connectors of three different performance classes.
- Glass materials including potash-lime-silica glasses M1 (sensitive) and M3 representative of medieval stained glass windows.
- Polymeric materials including polyamide and polyethylene.

As the most extensive materials exposure programme, the results of ICP Materials not only confirm the corrosive effect of SO_2 but also enable quantification for a wide range of materials. For most unsheltered materials also the effect of wet deposition (acid precipitation) has been quantified and comprises the second most important contribution to the corrosion rate. For selected materials the effect of ozone (O_3) and (NO_2) have been demonstrated.

ICP Materials is still going on. A finalised part, on which the present description is based, is the extensive 8-year field exposure programme that was started in September 1987 and involved 39 exposure sites in 12 European countries and in the United States and Canada. Dose-response functions have been obtained for many of the materials included, some of which are described in the following. Table 6.7 shows all parameters included in the final dose-response relations. In general, care should be taken when extrapolating the equations outside the range of environmental parameters used for their calculation.

Table 6.7 Parameters used in final ICP Materials D-R functions incl. symbol, description, interval measured in the programme and unit. All parameters expressed as annual averages.

Symbol	Description	Interval	Unit
T	Time	1-8	year
T	Temperature	2-19	°C
Rh	Relative humidity	56-86	%
$[SO_2]$	SO_2 concentration	1-83	$\mu g/m^3$
Rain	Rainfall	327-2144	mm
$[H^+]$	H^+ concentration	0.0006-0.13	mg/l

Stone Materials

Two stone materials have been exposed within ICP Materials, Portland limestone and white Mansfield sandstone. The function obtained for limestone is

$$R = (2.7\ [SO_2]^{0.48}\ e^{-0.018\ T} + 0.019\ \text{Rain}\ [H^+]) \cdot t^{0.96} \tag{6.16}$$

where R is the surface recession in µm, $[SO_2]$ is the SO_2 concentration in $\mu g/m^3$, T is the temperature in °C, t is the time in years, Rain is the amount of precipitation in mm/year and $[H^+]$ is the hydrogen ion concentration in precipitation in mg/l. The dose-response function was based on results from 100 observations and the R^2 value was 0.88. When performing cost calculations the maintenance frequency (1/t) is sought for a specified value of the surface recession, corresponding to a critical damage where maintenance action is required or desirable. Transforming Eq. (6.16) gives the function

$$1/t = [\ (2.7\ [SO_2]^{0.48}\ e^{-0.018\ T} + 0.019\ \text{Rain}\ [H^+])\ /\ R]^{1/0.96} \tag{6.17}$$

For sandstone, the damage function is slightly more complicated

$$1/t = [\ (2.0\ [SO_2]^{0.52}\ e^{f(T)} + 0.028\ \text{Rain}\ [H^+])\ /\ R]^{1/0.91} \tag{6.18}$$

where R is the surface recession in µm, $[SO_2]$ is the SO_2 concentration in $\mu g/m^3$, f(T) is a function of temperature in °C, equal to 0 when T is lower than 10°C and -0.013(T-10) when T is higher than 10°C, t is the time in years, Rain is the amount of precipitation in mm/year and $[H^+]$ is the hydrogen ion concentration in precipitation in mg/l. The dose-response function (R as a function of t and other parameters) was based on results from 101 observations and the R^2 value was 0.86.

It is in principle also possible to use the sandstone equation for other stone materials like rendering and mortar, however, with a higher degree of uncertainty, and probably underestimating the maintenance frequency. To apply it without modification to more resistant materials like granite or gneiss is not recommended.

Zinc and Galvanised Steel
Zinc is one of several metals that were exposed in the UN/ECE materials programme and the dose-response function from this programme proposed after 8 years of exposure is

$$ML = 1.4\,[SO_2]^{0.22}\,e^{0.018\,Rh}\,e^{f(T)}\,t^{0.85} + 0.029\,Rain\,[H^+]\,t \quad (6.19)$$

where ML is the mass loss in g/m^2, $[SO_2]$ is the SO_2 concentration in $\mu g/m^3$, Rh is the relative humidity in %, f(T) is a function of temperature in °C, equal to 0.062(T-10) when T is lower than 10°C and -0.021(T-10) when T is higher than 10°C, t is the time in years, Rain is the amount of precipitation in mm/year and $[H^+]$ is the hydrogen ion concentration in precipitation in mg/l. The dose-response function was based on results from 98 observations and the R^2 value was 0.84. For mathematical reasons it is not possible to transform Eq. (6.19) into an exact function that expresses 1/t as a function of all other variables. Instead, an estimate of 1/t ($1/t_e$) is proposed,

$$1/t_e = 0.14\,[SO_2]^{0.26}\,e^{0.021\,Rh}\,e^{f(T)} / R^{1.18} + 0.0041\,Rain\,[H^+] / R \quad (6.20)$$

where f(T) is a function of temperature in °C, equal to 0.073(T-10) when T is lower than 10°C and -0.025(T-10) when T is higher than 10°C. Eq. (6.20) has also been expressed in terms of thickness values, R (µm), instead of ML values (g/m^2) using the density for zinc (7.14). The reason for this is that the critical damage can be tied to zinc layer thickness values on different galvanised products. The estimate $1/t_e$ is always lower than the true value 1/t and the relative error never exceeds 6%. Formally, 1/t is bound by the conditions

$$1/t_e < 1/t \leq 1.06 / t_e \quad (6.21)$$

Paint Coatings
Both paint coatings on steel and galvanised steel were exposed within ICP Materials. The function for steel panel with alkyd is

$$(10\text{-}ASTM) = (0.033\,[SO_2] + 0.013\,Rh + f(T) + 0.0013\,Rain)t^{0.41} \quad (6.22)$$

where ASTM is the degradation measured according to ASTM D 1150-55, 1987 giving a rating between 1 and 10 where 10 corresponds to an unexposed sample, $[SO_2]$ is the SO_2 concentration in $\mu g/m^3$, Rh is the relative humidity in %, f(T) is a function of temperature in °C, equal to 0.015(T-11) when T is lower than 11°C and -0.15(T-11) when T is higher than 11°C, t is the time in years, Rain is the amount of precipitation in mm/year and $[H^+]$ is the hydrogen ion concentration in precipitation in mg/l. The dose-response function was based on results from 139 observations and the R^2 value was 0.68. The function can be transformed into a useable damage function by using the established criterion that maintenance should occur when ASTM=5,

$$1/t = [\ (0.033\ [SO_2] + 0.013\ Rh + f(T) + 0.0013\ Rain)\ /\ 5]^{1/0.41} \quad (6.23)$$

Similarly, the function for coil-coated galvanised steel with alkyd melamine is

$$1/t = [\ (0.0084\ [SO_2] + 0.015\ Rh + f(T) + 0.00082\ Rain)\ /\ 5]^{1/0.43} \quad (6.24)$$

where f(T) is a function of temperature in °C, equal to 0.040(T-10) when T is lower than 10°C and -0.064(T-10) when T is higher than10 °C. The dose-response function was based on results from 138 observations and the R^2 value was 0.73.

For carbonate-based paint coatings, a function is applied which was derived by Haynie (1986). Loss in thickness of the paint coating is estimated as

$$R = 0.12\ (1 - e^{-0.121Rh/(100-Rh)})[SO_2] + 0.0174Rain[H^+] \quad (6.25)$$

and the maintenance frequency, accordingly, to

$$1/t = (0.12\ (1 - e^{-0.121Rh/(100-Rh)})[SO_2] + 0.0174Rain[H^+])/R_{crit} \quad (6.26)$$

with Rh representing the relative humidity in %, Rain the annual precipitation in mm/a, $[H^+]$ the hydrogen ion concentration in precipitation in mg/l, and R_{crit} the critical surface recession for which country-specific values are applied.

Soiling of Building Materials

Soiling is the effect of particle deposition that results in a darkening of the surface and can be measured as a change in light reflectance. Dirty buildings are a common occurrence in all larger towns. Old churches are often almost black, although they were built of stones with fairly bright colours. Soiling is not only restricted to old buildings, greenhouses need to be cleaned, or solar cells have less output due to soiling. The alteration of the visual appearance may be unacceptable even if the base material is virtually unaffected and the costs related to cleaning may be substantial. Furthermore, the substances constituting the soiling matter (carbon particles), may indirectly take part in the degradation process by acting as a catalyst for various chemical reactions, particularly for the conversion of SO_2 and NO_x into sulphuric and nitric acids (Newby *et al*., 1991).

Soiling Mechanisms

The deposition of particles is characterised by the deposition velocity, similar to gas deposition. Particles deposit due to sedimentation, impaction and diffusion depending on the size of the particles. Sedimentation is important for particles larger than a few micrometres and mainly occurs at horizontal surfaces. Particles in this range have a high deposition velocity (typically 1mm/s). Since the lifetime of these particles in the atmosphere is short, they are found on surfaces near the source. Particles smaller than a few micrometres, i.e., sub-micron particles, have a much lower deposition velocity (typically 0.05 mm/s) thus less particles will reach the surface. Since diffusion is the main factor, the deposition takes place at any surface. Furthermore, sub-micron particles contain soot, therefore although a less effective deposition occurs, the effect is easily visible by the dirty appearance of the surface. Particles normally increase but may in some cases decrease the deterioration rates of materials, e.g., basic particles deposited on a surface may neutralise the effect of other pollutants, such as SO_2 (Tidblad and Kucera, 1998). This section will describe the quantification of soiling, which is only one effect that particles have on the degradation of materials.

Dose-Response Functions

All soiling dose-response functions include the concentration of particles in μg/m^3 as an explanatory variable. Since particles as such have a large variation in size and composition, the particle concentration depends on the analytical technique. The following parameters have been used in the quantification of soiling (QUARG, 1993):

- Dark smoke (DS) refers to non-reflective particulate matter measured by the smoke stain measurement method. The technique measures the reflectance of particulate matter collected on a filter paper compared to that of a calibration curve.
- Total suspended particulate (TSP) is measured as the weight increase of a filter paper collecting particles through a high volume air sampler. Although correlation factors have been given between DS and TSP, dark smoke measurements are, nowadays, not in any meaningful way comparable with gravimetric.
- Particulate elemental carbon (PEC) is also known as black carbon and graphitic carbon. Few measurements have been made of PEC levels. To identify the sources in the urban aerosol one possibility is to use PEC emission factors.
- PM_{10} refers to particular matter less than 10 μm aerodynamic diameter.

All these parameters have been used in dose-response functions but TSP and PEC are dominating. The preferred parameter can depend on the type of local pollution. The parameters mentioned here are only a list of parameters used so far and this does not exclude that other parameters are valuable for quantification of soiling effects.

The available dose-response functions are based on two types of models, the exponential model and the square root model.

The exponential model has a theoretical foundation. It expresses the reflectance, R, as

$$R = R_0 \cdot \exp\{-k_e \cdot C \cdot t\} \tag{6.27}$$

where R_0 is the reflectance of an unexposed surface, normally set to 100%, k_e is a constant, C is the particulate concentration in μg/m^3 and t is the time of exposure in years. Two different approaches both result in this form. The first approach, developed by Haynie (1986b), is based on the assumption that the loss of reflectance is caused by the deposition of a monolayer of opaque particles over the surface and that the loss is directly proportional to the fraction of covered surface. Lanting (1986) considered an alternative approach and described the loss of reflectance in terms of the thickness of the deposited film. Table 6.8 shows a summary of exponential dose-response functions. In addition to the theoretical functions only one author reports his data in this form.

Table 6.8 Summary of dose-response functions based on the exponential model, Eq. (6.27).

k_e	Particle	Surface	Reference
0.0085	TSP[a]	Theoretical[c]	(Haynie, 1986)
0.095	PEC[b]	Theoretical[c]	(Lanting, 1986)
0.13	PEC[b]	Theoretical[d]	(Lanting, 1986)
0.0092	TSP[a]	white-painted wood in a road tunnel	(Hamilton and Mansfield, 1993)
0.062	PEC[b]	white-painted wood in a road tunnel	(Hamilton and Mansfield, 1993)

a Total suspended particles
b Particulate elemental carbon
c Fraction of covered surface model
d Multilayer model

The square root model has only an empirical background and was first reported by Beloin and Haynie (1975), who gave the following expression for the reflectance, R,

$$R = R_0 - k_s (C \cdot t)^{1/2} \tag{6.28}$$

where R_0 is the reflectance of an unexposed surface, normally set to 100%, k_s is a constant, C is the particulate concentration in μg/m^3 and t is the time of exposure in years. Butlin *et al*.,(1994) report functions for many materials, including coated yellow brick

$$R = 43.21 - 0.1133 (C \cdot t)^{1/2} \tag{6.29}$$

where t is measured in months and C is expressed as total suspended particulate. Recalculating Eq.(6.29) for time measured in years and expressing the reflectance relative to that of the original gives

$$R = 43.21 - 0.39 (C \cdot t)^{1/2} \approx 43 (100 - 0.91 (C \cdot t)^{1/2}) \tag{6.30}$$

Table 6.9 shows a summary of square root dose-response functions. Note especially that for Eq.(6.30) the compiled value is 0.91, i.e. that which gives the reflectance normalised to 100%.

Almost all functions given in Table 6.9 have the particle concentration expressed as TSP (total suspended particles). Exceptions are those of Pio *et al*. (1998) which are based on concentrations measured as PEC (particulate elemental carbon). In these cases alternative functions have been calculated based on the original functions and particulate measurements of TSP and PEC tabulated in Pio *et al*. (1998).

Table 6.9 Summary of dose-response functions based on the square root model, Eq.(6.28).

k_s	Particle	Surface		Reference
1.4	TSP[a]	Acrylic emulsion paint	unsheltered	(Beloin and Haynie, 1975)
2.8	TSP[a]	Painted wood	unsheltered	(Hamilton and Mansfield, 1993)
4.2	TSP[a]	Painted wood	sheltered	(Hamilton and Mansfield, 1993)
1.1[c]	TSP[a]	Oil based paint	unsheltered	(Butlin *et al*., 1994)
1.1[c]	TSP[a]	Tint based paint	unsheltered	(Butlin *et al*., 1994)
2.2[c]	TSP[a]	Acrylic emulsion paint	sheltered	(Butlin *et al*., 1994)
1.6[c]	TSP[a]	Acrylic emulsion paint	unsheltered	(Butlin *et al*., 1994)
1.6[c]	TSP[a]	Shingles	unsheltered	(Butlin *et al*., 1994)
0.91[c]	TSP[a]	Coated yellow brick	unsheltered	(Butlin *et al*., 1994)
4.5	PEC[b]	Painted wood	sheltered	(Pio *et al*., 1998)
5.7	PEC[b]	Portland stone	sheltered	(Pio *et al*., 1998)
1.3	TSP[a,d]	Painted wood	sheltered	(Pio *et al*., 1998)
1.6	TSP[a,d]	Portland stone	sheltered	(Pio *et al*., 1998)

[a] Total suspended particles; [b] Particulate elemental carbon; [c] The values have been adjusted to R_0=100%; [d] The values were recalculated from a PEC equation using data measured at the site.

Both, the exponential model (6.27) and the square root model (6.28) can be transformed into exposure-response functions,

$$1/t = C \cdot k_e / \ln(R_0 / R) \tag{6.31}$$

and

$$1/t = C \cdot k_s^2 / (R_0 - R)^2 \tag{6.32}$$

respectively. Eqs. (6.31) and (6.32) both have the general form

$$1/t = C / d_{crit} \tag{6.33}$$

where d_{crit} is the 'critical' dose $(C{\cdot}t)_{crit}$. In other words, after application of a suitable maintenance criterion $R=R_{crit}$, both functions result in the same damage function. According to Hamilton and Mansfield (1993) and Pio *et al*. (1998) maintenance is triggered when R_{crit}=70%. Using the data of Pio *et al*. (1998), and this criterion, the critical soiling dose for sheltered painted wood and limestone due to soiling becomes 533 and 352 year·µgTSP/m^3, respectively. For sheltered painted wood the Hamilton and Mansfield (1993) data gives an alternative critical dose of 51 year·µgTSP/m^3, i.e. ten times lower. In view of the longer exposure period, 850 vs. 120 days, the Pio data is preferred, recognising however, that it may overestimate the maintenance time, i.e. underestimate the replacement frequency.

On the other hand, in areas with high particulate matter levels, individuals can possibly accept greater reductions in reflectance before cleaning due to the general impression of dirty buildings. When people judge the soiling status of an object, they do so compared to a surface in the surroundings, which is considered to be white. In reality, this white surface may also be soiled to a lesser extent depending on the general pollution level. Therefore, the maintenance criterion R=70% may be modified.

Table 6.10 shows the result of these modified criterions, calculated for the exponential and square root model and based on two assumptions, one that can be considered reasonnable ('almost white') and one extreme ('grey'). In practice, this means that the critical doses mentioned can also be higher than what can be expected when using R=70% and the Pio functions.

Table 6.10 Calculated critical doses (see Eq. (6.33)) for polluted areas based on a higher tolerance to soiling due to individuals perceiving partially soiled surfaces as white.

Surface perceived as white is	Critical dose	
	Exponential model	Square root model
White (reference case[a])	$d_{crit} = \ln(1 / 0.7) / k_e$	$d_{crit} = 900 / k_s^2$
Almost white (10-percentile of soiled objects)	1.1 d_{crit}	1.6 d_{crit}
Grey (average soiling level)	1.9 d_{crit}	3.5 d_{crit}

[a] See Table 6.8 and Table 6.9 for k_e and k_s values, respectively.

Combination of Degradation and Soiling Effects

The exposure conditions are of great importance when estimating impacts. Horizontal surfaces soil more rapidly than vertical surfaces. Sedimentation and impaction apply for particles larger than 1 µm and are orientation-dependent whereas convective diffusion is the mechanism for deposition of sub-micron particles. Also, wind and rain easily remove coarse (> approximately 2 µm) particles while fine (< approximately 2 µm) particles adhere more strongly (Creighton *et al*., 1988). In a soiling study performed over a relatively long time (850 days), the unsheltered data was erratic. The study included soiling measurements on painted wood and stone tablets and also corrosion measurements on the stone tablets. On unsheltered painted wood, the soiling was initially rapid but, for longer exposures, there was a cleaning effect and the reflectance increased with time. The unsheltered stone tablets showed an increase in reflectance and later a decrease, however, not reaching below the initial value. When attempting to correlate these changes with rain events, periods of agreement between observed and predicted values were as common as the opposite behaviour. On many occasions the rain does not remove the deposited particles but only rearranges them on the surface. Dose-response functions were obtained for sheltered samples but not for unsheltered samples. Measurements of corrosion attack revealed that the unsheltered tablets decreased in weight whereas sheltered tablets increased in weight (Pio *et al.*, 1998). The latter finding is consistent with observations within the ICP Materials programme. The weight increases for sheltered stone samples are due to a combination of soiling and corrosion effects. In summary, it is possible to obtain dose-response functions for corrosion attack on unsheltered stones and paint coatings and soiling of sheltered samples. For corrosion of stone materials in sheltered positions and for soiling of unsheltered objects, however, functions are at the present stage not reliable.

Any building is subject to impact both by deterioration and by soiling and the decision to take a maintenance action may be due to deterioration or soiling or both. Different maintenance practices need to be considered in order to combine the results obtained from calculations on degradation and soiling of materials. Figure 6.2 shows an illustration of selected practices for a painted surface. In the first case no action is taken and corrosion occurs after a time, ta, with an associated maintenance frequency, $1/t_a$. In the second case cleaning is performed at regular intervals, which prolongs the time to corrosion, thereby reducing the cost. On the other hand, the cleaning cost at frequency $1/t_b$ is added to the total cost. In the last case repainting and cleaning is performed at regular intervals, practically eliminating the cost due to corrosion. Each of the practices a, b and c may be the most cost effective approach, depending on the material and application. In reality the strategy used may also be a mix of the three approaches.

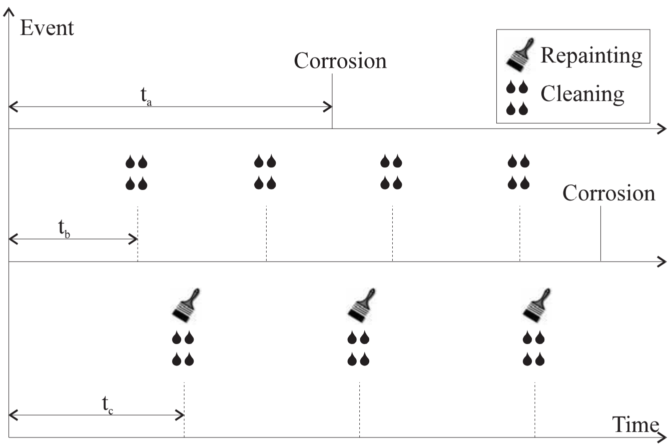

Figure 6.2 Illustration of different maintenance practices showing a) no action, top b) cleaning at regular intervals, middle, and c) cleaning and repainting at regular intervals, bottom. For further discussion see text.

In view of this, how should the maintenance frequencies, and associated costs, calculated from the degradation functions (6.17), (6.18), (6.20), and (6.24), on the one hand, and the critical soiling doses, on the other hand, be interpreted? The degradation functions are valid for unsheltered surfaces while the critical soiling doses are valid for sheltered surfaces. At the present stage, however, the data on stock of materials at risk can not be divided into fractions corresponding to unsheltered and sheltered surfaces. Consider, for example, a facade consisting of partly unsheltered and partly sheltered areas. The decision to clean/paint may be taken either due to unacceptably high corrosion in unsheltered areas or due to unacceptably high soiling of sheltered areas. Once the decision is taken, however, it is likely that the entire facade is cleaned/repainted regardless of the sheltering status of individual areas. Therefore the cost associated with the action should be multiplied by the total stock at risk area. The resulting total cost therefore represents one of two extreme cases. In the first it is assumed that all maintenance actions are taken as a direct result of corrosion damage while in the second it is assumed that all maintenance actions are taken as direct results of soiling. It is not possible at the present stage to combine these cost estimates. To conclude, the following observations should be taken into account:

- New dose-response functions are available from ICP Materials, an international exposure programme, describing the degradation of a wide range of materials including metals, stone materials, paint coatings, electric contact materials, glass materials and polymeric materials.
- Selected functions from the programme, valid for unsheltered positions, have been adapted and exposure-response functions suitable for inclusion into EcoSense have been derived for limestone (Eq.(6.17)), sandstone (Eq. (6.18)), zinc/galvanised steel (Eq. (6.20)), painted steel (Eq.(6.23)) and painted galvanised steel (Eq. (6.24)). They include SO_2, temperature and relative humidity for quantification of dry deposition effects, and amount and acidity of precipitation for quantification of wet deposition effects.
- A critical review of dose-response functions for soiling to materials has been performed including additional analysis on strategies for selecting maintenance intervals. The functions quantify the loss of reflectance in terms of the particulate concentration times time of exposure, i.e. the particulate dose, which means that each exposure-response function can be summarised in the form of a critical dose. No other characteristics of the environment are included in the soiling functions.
- Critical soiling doses in sheltered conditions have been estimated to be 533 and 352 year·μgTSP/m^3 for painted wood and limestone, respectively (TSP = total suspended particulate). The uncertainties of these critical doses are high compared to the degradation functions and can be either significantly lower or higher.
- It is not possible today to combine cost estimates related to degradation and soiling of materials into a single cost estimate representative for the total impact to materials. Instead, the individual estimates should be regarded as illustrations pertaining to particular isolated maintenance practices.

6.7.2 Impacts on crops

This section draws on the latest methodological developments within the ExternE-Pol project (Int Panis *et al*., 2004) and on earlier reports of the ExternE methodology (European Commission, 1995 and 1999).

Effects from SO_2

The function for effects from SO_2 recommended in ExternE is adapted from one derived by Baker *et al*. (1986). The function assumes that yield will increase with SO_2 from 0 to 6.8 ppb, and decline thereafter. The function is used to quantify changes in crop yield for wheat, barley, potato, sugar beet and oats and is defined as

$$y = 0.74 \cdot [SO_2] - 0.55 \cdot [SO_2]^2 \qquad \text{for } 0 < [SO_2] < 13.6 \text{ ppb} \tag{6.34}$$

$$y = -0.69 \cdot [SO_2] + 9.35 \qquad \text{for } [SO_2] > 13.6 \text{ ppb} \tag{6.35}$$

with y = relative yield change
$[SO_2]$ = SO_2-concentration in ppb

Effects from Ozone

For the assessment of ozone impacts, a linear relation between yield loss and the AOT 40 value (Accumulated Ozone concentration above a Threshold of 40 ppbV) calculated for the growth period of crops (May to June) is assumed (Fuhrer 1996, Mills *et al*., 2003). The relative yield change is calculated using the following equation together with the sensitivity factors given in Table 6.11:

$$y = 99.7 - \alpha \cdot AOT40_{crops} \tag{6.36}$$

with y = relative yield change
α = sensitivity factors

Table 6.11 Sensitivity factors for different crop species.

Crop species	Sensitivity factor α
Rice	0.4
Tobacco	0.5
Sugar Beet, potato	0.6
Sunflower	1.2
Wheat	1.7

Acidification of Agricultural Soils

An upper-bound estimate of the amount of lime required to balance atmospheric acid inputs on agricultural soils across Europe is estimated. Ideally, the analysis of liming would be restricted to non-calcareous soils, but this refinement has not been introduced given that even the upper-bound estimate of additional liming needs is small compared to other externalities. The additional lime required is calculated as:

$$\Delta L = 50 \text{ kg/meq} \cdot A \cdot \Delta D_A \tag{6.37}$$

with ΔL = additional lime requirement in kg/year
A = agricultural area in ha
ΔD_A = annual acid deposition in meq/m^2/year

Fertilisation Effects from Nitrogen Deposition
Nitrogen is an essential plant nutrient, applied by farmers in large quantity to their crops. The deposition of oxidised nitrogen to agricultural soils is thus beneficial (assuming that the dosage of any fertiliser applied by the farmer is not excessive). The reduction in fertiliser requirement is calculated as:

$$\Delta F = 14.0067 \text{ g/mol} \cdot A \cdot \Delta D_N \tag{6.38}$$

with ΔF = reduction in fertiliser requirement in kg/year
A = agricultural area in km^2
ΔD_N = annual nitrogen deposition in meq/m^2/year

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7 Impact Pathway Approach: Monetary Valuation

7.1 Overview of General Approach

As interest in external costs associated with energy and transport increases so the need for refinement and elaboration of the ExternE methodology continues. This chapter reports on recent developments in the monetary valuation part of the overall external cost estimates. More specifically, this chapter documents how monetary values have been up-dated for the following impact categories:

- Health
- Noise
- Buildings, visibility and transmission lines
- Crops

7.2 Valuation of Health Impacts of Air Pollution

7.2.1 Mortality

The objective of this section is the derivation of unit values to account in monetary terms for the incidence of premature death, estimated to result from air pollution in Europe. Values were derived from three surveys undertaken simultaneously in UK, France and Italy, using a common survey instrument.

The impact-pathway approach to the estimation of environmental external costs adopted in the European Commission-funded ExternE research project requires – for its completion – the monetisation of the impact end-points identified by the modelling of pollution effects[7] arising from energy and transport fuel cycles. In the case of air pollution, the epidemiological literature presented in previous phases of ExternE has signalled that exposure to a number of pollutants, including particulates, nitrates, sulphates and ozone, (e.g. European Commission, 1999), can lead to cases of immediate (acute) or delayed (chronic) premature death within a given population. There is therefore the need for a unit value to represent each estimated instance of premature death in the final estimation of environmental external costs.

The search for appropriate unit values has until now relied on the available literature. However, as explained in further detail below, the values that currently exist are generally not believed to represent accurately the willingness to pay (WTP) that individuals might express, e.g. for the introduction of a new air quality regulation. More specifically, existing values are derived often in the context of the workplace

[7] See chapter 4, European Commission (1995 and 1999) for details of the impact pathway methodology.

(wage-risk studies) that estimate the willingness to accept (WTA) a higher wage rate in accordance with a greater risk of accidental death. Alternatively, attention has been given to the valuation of fatal transport accidents, the frequency of which might be expected to change with the introduction of new transport infrastructure for example.

Both the road and workplace examples of contexts differ from the context of air pollution and so may be expected to result in different WTP values. The principal differences are:

- The length of lifetime lost on average through the impact. Whereas the impact of premature death in the road or work context can be expected to be on an individual of average age within the population and therefore result in the loss of about 35 years of life, air quality impacts are typically likely to lead to a loss of life of only a few weeks or months.
- The state of health of the individual impacted. Whilst the epidemiological literature suggests that air-pollution death is more likely to result in the case of an individual who has an already-existing impaired health condition, the typical victim of a premature death in the road or work context can be expected to be in good health.

There are a number of other potentially important differences between the contexts that might therefore lead to different WTP values. These are:

- Size of the risk change. It has been suggested that the annual risk change associated with a realistic air pollution policy may be 10^{-4} whilst the risk valued in the transport accident context is typically 10^{-3}.
- Context specificity. The nature of the risk is perceived to be different according to the degree to which exposure to the risk is voluntary, the extent to which the potential impact is perceived to be controllable, and the size of the impact (in terms of number of deaths resulting). For example, premature death as a result of a road accident is likely to be perceived to be more voluntary than a death that results from ambient air pollution.
- Immediacy of the impact. Premature death resulting from a transport or workplace context is likely to result immediately following an accident. Conversely, there is often a lapse of time between being exposed to air pollution and feeling the health effects – that is, the effects are latent.

These differences give rise to the possibility that the unit values that should be applied to the air pollution external cost estimation differ from those derived in other contexts. For a long time the ExternE team has been constrained to adopt such values and then adjust them to account for these differences, as far as theory and evidence allow. In practice, the main adaptation of the unit values derived from wage-risk (and other) studies has been to try to account for the length of lifetime lost by changing the metric from the VPF (value of a prevented fatality, also called VSL = value of a statistical life), to the VOLY (value of life years)[8].

Outlining the differences in context from where the values are derived (wage risk, consumer markets etc.) and where they are used (air pollution), as we do in the next section, indicates that there are reasonable grounds to expect that the unit values need not be the same. This provides the principal justification for the present study which tries to derive unit values that are more appropriate and reliable in policy use.

The need for reliability in policy analysis as a motivator for the current study is underscored when it is remembered that in previous ExternE analyses health impacts comprise 98% of the external costs from SO_2 and 100% of those from particulates (European Commission, 1999), with mortality impacts accounting for at least 80% of these health impacts. Since this impact-pathway is critical to the scale of the external cost estimates, it is important that the individual components of the pathway are as robust as possible.

General Methodological Issues

Hunt *et al*. (2004) provides a literature review of the approaches and empirical methods used to estimate the value of a statistical life. Two general approaches have been used for the valuation of the benefits of life-saving activities, including environmental programmes that reduce risks of death: the human capital approach and the willingness to pay approach (e.g. Cropper and Freeman, 1991; Berger *et al.,* 1994; Johansson, 1995). The first approach measures the economic productivity of the individual whose life is at risk. It uses an individual's discounted lifetime earnings as its measure of value, assigning valuations in direct proportion to income. The Willingness to Pay approach has its basis in the assumption that changes in individuals' economic welfare can be valued according to what they are willing (and able) to pay to achieve that change. According to this assumption, individuals treat longevity like any consumption good and reveal their preferences through the choices that involve changes in the risk of death and other economic goods whose values can be measured in monetary terms.

Various methods have been used in order to make empirical estimations of willingness to pay, each providing a means to derive Hicksian measures for individuals making trade-offs between risks to life and health and other consumption goods and services. We focus our attention on three methods outlined below. These are: the Compensating Wage, the Averting Behaviour and the Contingent Valuation methods.

Compensating Wage Method

To date, the compensating wage method has been the predominant empirical approach to assess willingness to pay for reductions in the risk of premature death. The method

[8] Friedrich and Bickel (2001).

uses labour market data on wage differentials for jobs with health risks and assumes that workers understand very well the workplace risk involved and that the additional wage that workers receive when they undertake risky positions reflects risk choice. Compensating wage models are consistent with the willingness-to-pay approach in the sense that they recognise that individuals have unique preferences over risky alternatives and that they have opportunities to reduce risks, depending on their labour skills. These models postulate that part of the differences in risk preferences are systematic and depend on objective and measurable individual characteristics. However, "…much of the criticism of the Compensating Wage approach centres on its assumptions concerning the labour market. Many critics argue that the actual labour market bears little resemblance to the labour market described in Compensating Wage models. The compensating wage approach assumes that workers are fully cognisant of the extent and consequences of the on-the-job risks they face, that labour market is strictly competitive, and that insurance markets are actuarially correct, with premiums and payouts matched to accurately assessed risks" (Kuchler and Golan, 1999).

A recent study by Viscusi and Aldy (2003) reviews a large number of more recent wage-risk studies. The European studies – mostly from the UK – are summarised in Table 7.1 below.

Table 7.1 Summary of European labour market studies of the VSL.

Author (year)	Country	Annual Mean risk	Implicit VSL (€ million, 2000 prices)
Marin and Psacharopoulos (1982)	UK	0.0001	4.3
Weiss, Maier and Gerking (1986)	Austria	n.a.	4.0 – 6.6
Siebert and Wei (1994)	UK	0.000038	9.5 – 11.6
Sandy and Elliot (1996)	UK	0.000045	5.3 – 69.6
Arabsheibani and Martin (2000)	UK	0.00005	20.0
Sandy, Elliot, Siebert and Wie (2001)	UK	0.000038	5.8 – 74.4

Source: Viscusi and Aldy (2003)

The range of values generated by these studies is a little disconcerting and reflects the different model specifications used. A conservative mean value of VPF from the lower end of these ranges is around €5 million. A meta-analysis of 17 studies by CSERGE (1999) generated a range of VPF between €2.9 million and €100 million. The weighted (by sample size) arithmetic mean, when biases introduced by sample data and the analytical approach were controlled, was €6.5 million (2002 prices).

The applicability of these results in the context of air pollution is questionable – most obviously by the fact that the compensating wage method estimates the value of a statistical life based on information about the labour market, where old people are generally absent. Since older people have fewer life-years remaining than young people, the compensation received in labour market studies may overstate the value of risk reductions to old people, for whom the risk of premature death appears to be most relevant. The health condition of these two groups is also likely to differ significantly. Additionally, the context is very different: wage risk trade-offs are assumed to be voluntary whilst the air pollution context is a more involuntary one.

The Avertive Behaviour Method

The avertive behaviour method assumes that individuals spend money on certain activities that reduce their risk of death, like buying smoke detectors or seatbelts, and that these activities are pursued to the point where their marginal cost equals their marginal value of reduced risk of death. The marginal costs incurred by individuals to reduce their probability of death is used to value individuals' willingness to pay to reduce their risk of death. Given individual data on the marginal costs of an averting good, the willingness to pay for avoiding premature death can be estimated.

The relevant measure of the effect of the averting behaviour on risk of death is, according to Cropper and Freeman (1991), the individual's perception of this risk reduction. Although relevant, these perceptions are difficult to observe and data are hard to come by.

Evidence (e.g. Viscusi, 1993; European Commission, 1999) suggests that the conclusion of Cropper and Freeman (1991) is likely to hold in practice. Average VPFs of €1-1.5 million are found in these studies. Whilst it is possible to link air pollution incidence with consumer expenditure (e.g. on housing) it has proved very difficult to relate such behaviour specifically with the risk of premature death, and to separate it from morbidity effects (see Klemmer *et al.*, 1994 for a discussion of the evidence).

Contingent Valuation Method (CVM)

Contingent Valuation is a survey method in which respondents are asked to state their preferences in hypothetical, or contingent, markets, allowing analysts to estimate demands for goods or services that are not traded in markets. The CVM draws on a sample of individuals who are asked to imagine that there is a market where they can buy the good or service evaluated, stating their individual willingness to pay for a change in the provision of the good or service, or their minimum compensation (willingness to accept) if the change is not carried out. Socio-economic characteristics of the respondents – gender, age, income, education, etc. – and demographic information are obtained as well. If it can be shown that individuals' preferences are not random, and instead vary systematically and relate to some observable demographic characteristics, then population information can be used to forecast the aggregate willingness to pay for the good or service evaluated.

There is a large body of knowledge on the method's advantages and disadvantages (e.g. Mitchell and Carson, 1989). The main advantage – as implied above – is that the CVM can estimate a WTP for a good/service for which there are no market data. The central problem in a contingent valuation study is to make the scenario sufficiently understandable, clear and meaningful to respondents, who must understand clearly the changes in characteristics of the good or service he or she is being asked to value. The mechanism for providing the good or service must also seem plausible in order to avoid scepticism that the good or service will be provided, or that the changes in characteristics will occur.

The applicability of the contingent valuation method in the air pollution context appears to be high since the survey instrument allows the researcher to relate the WTP question precisely to the nature of the commodity to be valued – something that is not so easily possible in the market-based approaches. Its success therefore is determined by how effectively the survey instrument minimises the biases listed above. Most importantly, the scenario elements of the hypothetical market in the survey instrument must be understandable, meaningful and plausible to respondents.
In this section, we give a brief review of evidence based on CVM studies that relate to our search for unit values in the air pollution context, and in particular the issues of age, health status and context. The first study to address the issue of age dependency of VPFs was by Jones-Lee (1989), which examined individuals' WTP for reducing the risk of serious motor vehicle accidents. Based on a central VPF of €4 million at age 40, the age-VPF variance was found to have an inverted U-shape. Other supporting evidence for a pattern of VPF declining with age is found in Desaigues and Rabl (1995) and Krupnick *et al*. (2000) – the latter using the survey instrument adopted in the study described below, in the Canadian context.

A more recent study is that of Johannesson and Johansson (1996) who use the contingent valuation method to look at the WTP of different respondents aged 18-69 for a device that will increase life expectancy by one year at age 75. A sample of the results obtained is reported in Table 7.2.

Table 7.2 WTP (€ price year 2002) for one year of life at age 75 and corresponding values for one year of life immediately.

Age of Payment	WTP for 1 Life Year at 75	WTP for 1 Life Year Now (3% Discount rate)
18-34	1676	7176
25-51	2120	6327
52-69	2433	3733

Source: M. Johannesson and P-O Johansson (1996)

The Johannesson and Johansson results show an increasing WTP with age – although criticism has been levelled at this study on the basis of its elicitation method and small sample size. This pattern relating to age has also been found in a CVM study by Persson and Cedervall (1991). Pearce (1998) concludes on the basis of a review of the literature that the evidence, such that it is, seems to favour a case for a slow decline of VPF with age. The related issue of futurity of impact (from latent and chronic mortality air pollution effects) has, as far as we are aware, only been empirically estimated in the Alberini *et al*. studies in North America, (Alberini *et al.,* 2001). These studies show that future risk changes are valued lower than immediate risk changes in both the USA and Canada, resulting in internal discount rates of 4.6% and 8% respectively.

Regarding a relationship between *health status* and VPF, the CVM evidence is very limited and inconclusive. The principal studies that have explored this linkage are Johannesson and Johansson (1996) who found that WTP values declined with poorer health status, whilst Krupnick (2000) found no significant evidence of a relationship.

The relationship between WTP and *context* is similarly under-developed in terms of primary CVM studies. The main studies, by Jones-Lee and Loomes (1993, 1995) and Covey *et al.* (1995), reported in Rowlatt *et al.* (1998) consider the road transport accident VPF in relation to those for underground rail accident risks, food risks, risks to third parties living in the vicinity of major airports and domestic fire risks. The perceived involuntariness of the underground rail risk attracted a 50% premium on the road VPF, whilst a 25% discount is attached to the risk of a domestic fire. The latter result was thought to reflect the high degree of voluntariness or controllability in this context. No evidence was found to support an adjustment to the road accident VPF for scale of the accident (i.e. in the case of the contexts of underground accident or residents' proximity to airports). Thus, the limited evidence suggests context relating to voluntariness is likely to be important in determining WTP but the weight of evidence for this is not yet strong enough to draw this as a strong conclusion.

A point to be observed when using the Contingent Valuation method for eliciting the willingness to pay for a reduction in probabilities of death is how sensitive the estimates are to changes in risk. Economic theory suggests that willingness to pay to reduce small probabilities of death should be increasing with the magnitude of risk reduction, and be approximately proportional to this magnitude, assuming that risk reduction is a desired good. For example, if a reduction in annual mortality risk is valued at a certain amount of money, then a larger reduction in risk should be valued at a larger amount of money. In addition, the difference between the values should be proportional to the difference in risks, ignoring the income effect.

Hammitt and Graham (1999) discussed some reasons why stated willingness to pay often is not sensitive to variations in risk magnitude. One possible reason, they argued based on the review of several CVM studies, is that respondents might not understand probabilities or lack intuition for the changes in small probabilities of death risk. Another possibility relates to the fact that respondents might not treat the given probabilities as given to them. As a consequence, stated willingness to pay would not be proportional to the amount of risk reduction given to respondents, but should be proportional to changes in perceived risk.

In order to test for this, an internal test of sensitivity to magnitude, within a given sample, can be performed, where the respondent is asked for willingness to pay for different changes in risk in the same questionnaire. An 'external' test of sensitivity to magnitude occurs when different samples are used to compare the willingness to pay estimates, i.e. different respondents are asked about their willingness to pay for different risk reductions and there is no possibility of co-ordinating their responses. Internal tests are more likely to be successful because respondents are likely to base their responses to willingness to pay questions about one risk reduction on their answers to previous questions about a different risk change, anchoring their answers on their previous responses and enforcing some degree of consistency. Alberini *et al.* (2001) find that WTP for risk reductions varies significantly with the size of the reduction in the Canadian application of the present survey instrument. Mean WTP for an annual reduction in risk of death of 5 in 10,000 in this case was about 1.6 times WTP for an annual risk reduction of 1 in 10,000, showing sensitivity to the size of the risk reduction, but not strict proportionality.

Alternative Metrics

There has been considerable debate within the ExternE team as to whether the value of a prevented fatality (VPF) should be replaced by the value of life years (VOLY) as the principal metric by which to value incidence of premature death from air pollution. Table 7.3 below summarises some of this thinking. Rabl (2002) proposed a key argument in this debate. He shows that the number of deaths that can be attributed to this cause is only observable in mortality statistics when the exposure-death effect is sufficiently instantaneous that the initial increase in death rate is not obscured by the subsequent depletion of the population who would otherwise die later.

Rabl argues that the usual case is that the impact of air pollution is not instantaneous but is the cumulative result after years of exposure, so that the number of deaths is not observable[9]. As a result, it is impossible to tell whether a given exposure has resulted in a small number of people losing a large amount of life expectancy or a lot of people losing a small amount of life expectancy. In this case only the average number of years of life lost is calculable and so makes a strong case for the use of VOLYs in the context of air pollution.

[9] In this case, for example, affected individuals may die over a period of 30 years following exposure. Some individuals may die in the second year of this period who would have died anyway in year 20. But individuals may die in year 20 from the exposure. Any change in the observable mortality rate in year 20 therefore understates the true mortality rate that can be attributable to air pollution.

Table 7.3 Appropriateness of value metrics in different contexts.

Type of impact to be valued and evaluation criteria	VPF	VOLY	Conclusion
Instantaneous Δ in risk of death	WTA/WTP Δ Risk (R) Varies with age[1] Varies with Δ Risk size	WTA/WTP Δ Length of lifetime remaining (L) varies with age[2] may vary with L	No means to prefer one to the other
Change in latent risk or in risk probability profile	WTA/WTP Δ Risk (R) - Δ in future R valued on a discounted basis	WTA/WTP Δ Length of time (L) varies with age[2] may vary with size of L	Bias in favour of VOLY because: interpretation for empirical work is easier VPF equivalent is difficult to define
Valuation of time-delayed mortality – dose-response function gives loss of life years	Construct an artificial equivalent loss of lives and then apply VPF from other studies	Apply VOLY obtained from other studies including ExternE	Clear preference for VOLY
Valuation of accidental death	Apply VPF to Δ in probability of death	Apply VOLY times loss of life expectancy to get a value; multiply by Δ in probability of death	VPF may be easier to use.
Estimation of VOLY from VPF	No need	Assuming: constant discount rate simplistic relationship between VPF and life expectancy	Not recommended as way of obtaining VOLY.
Public acceptability	Very low in policy terms	May be little higher although scope for misunderstanding is still there	Marginal preference for VOLY
Confusion of *ex post* and *ex ante*	Common confusion in public mind	Perhaps less susceptible to wrong argument	Marginal preference for VOLY
Link to other measures	Cannot be linked to (e.g. health) policies that affect QUALYs	Link to QUALYs exists and can be developed	Preference for VOLY

[1] Theory and empirical evidence support an inverted U-shape but theory excludes value of survival and possibilities of changes in preferences for risk as we grow older. Moreover, empirical evidence is quite limited.
[2] Theory might suggest declining values with age (loss of life expectancy falls as you get older). But we still must allow for changes to attitudes to risk etc.

Justification of Research Methodology

The sections above have demonstrated that, in order to derive reliable unit values for the risk of premature death from exposure to air pollution, it is important to consider a number of factors including latency, age and health condition. These issues had previously been addressed in a survey instrument developed by Krupnick and colleagues at the Resources For the Future (RFF). The survey has been used in studies for the USA and Canada and results are reported in Alberini *et al.* (2001). It was decided by the ExternE team that it would be prudent in the first instance to adopt an existing survey instrument. Reasons included the facts that:

- Development costs could be minimised;
- in the course of its implementation in North America it had already been the subject of peer group review and represented the state-of-the-art;
- it allowed comparability with the North American results.

The structure of the survey instrument and key arguments relating to important design features, including the ways in which it attempts to address a number of biases associated with contingent valuation studies, are outlined in Hunt *et al*. (2004).

The survey in its current format has been developed over a period of several years using extensive face-to-face interviews in the USA, and has been pre-tested in the USA, Japan and in Canada. The survey instrument is designed to elicit WTP for mortality risk reductions to be incurred over 10 years (effective immediately) and for reductions in the probability of dying between age 70 and 80. It has been developed by the members of the project team and under the guidance of a cognitive psychologist, and has relied heavily on the use of the so-called "think-aloud" protocol to elicit "mental models" of risk perception and its relationship to willingness to pay. The development work for this instrument includes 30 personal interviews, eight focus groups, and two pre-tests involving a total of 80 people. The instrument has been developed in order to tackle problems, in particular insensitivity to the scope of the commodity that has been found in previous studies.

The survey instrument is self-administered and computerised, thereby removing any interviewer biases. The components of the survey are described in the order that they appear in a series of computer screens. The use of a series of tele-visual screens allows the graphics to be made clearer and more adaptable to the individual than would be possible with printed questionnaires. Comprehension is also improved by reinforcing the written text with voice-overs, so that respondents will both see and hear questions. This has shown to be particularly important in the case of older respondents. Experience in North America showed that the use of interactive screens, as opposed to face-to-face interviews for example, does not present a deterrent on "fear of technology" grounds and, in fact, facilitates the advantages mentioned above.

An example of the WTP questions is:

Suppose that a new product becomes available that, when used over the next ten years, would reduce your chance of dying from a disease or illness. This product would reduce your total chance of dying over the next ten years from X to Y.

If you were to take this product you would have to pay the full amount of the cost out of your own pocket each year for the next ten years. For the product to have its full effect, you would need to use it every year for all ten years.

We realise that most people will not simply accept the idea that this product is guaranteed to work without some proof. In answering the next questions, please assume that the product has been demonstrated to be safe and effective in tests required by the UK Government.

Keeping in mind that you would have less money to spend on other things, would you be willing to pay €Z per year (10 times Z total) to purchase this product?

Results of country studies and pooled analysis

We summarise the individual country studies and present the results from an econometric analysis that pools the data from the individual surveys. The latter analysis allows us to explore the possibility that unit values for the EU as a whole can be based on the survey data from a range of countries. Alternatively it allows us to speculate as to whether unit values in individual countries can be explained by observable variables e.g. income, or whether cultural differences render any such analysis and derivation of common unit values a fruitless exercise.

Respondents were shown their baseline risk of death over the next 10 years, which varies with gender and age, and were subsequently asked to report information about their WTP for (i) a risk reduction of 5 in 1000, to be incurred over the next 10 years, with respect to the baseline, and (ii) a risk reduction of 1 in 1000, to be incurred over the next 10 years, with respect to the baseline. In addition, respondents were told about their baseline risk of death at age 70 over the subsequent 10 years, and were queried about their WTP for (iii) a 5 in 1000 risk reduction, which would begin at age 70 and be spread over the next 10 years. The payment, respondents were told, would have to be made every year, and would begin immediately.

Attention is restricted to WTP for the 5-in-1000-risk reduction over the next 10 years and details of the studies can be obtained in Alberini *et al.* (2004).

To obtain estimates of mean and median WTP, we combine the responses to the initial and follow-up payment questions to form intervals around the respondent's (unobserved) WTP amount. We further assume that WTP follows the Weibull distribution with scale parameter σ and shape θ, and estimate these parameters using the method of maximum likelihood. The log likelihood function of the WTP data is:

$$\log L = \sum_{i=1}^{n} \log\left[\exp\left(-\left(\frac{WTP_i^L}{\sigma} \right)^{\theta} \right) - \exp\left(-\left(\frac{WTP_i^U}{\sigma} \right)^{\theta} \right) \right] \tag{7.1}$$

where WTP^L and WTP^U are the lower and upper bound of the interval around the respondent's WTP amount. Equation (7.1) describes an interval-data model. We first fit this model separately for the Italy, France and UK data, and then we consider pooled-data models.

We work with the Weibull distribution because WTP for a risk reduction should be non-negative. Other distributions, such as the lognormal, are suitable for non-negative variates, and indeed we did compare the fit of the Weibull with that of other distributions that do not admit negative values, including the lognormal, exponential and log-logistic. The fit of the Weibull was always better.

Another reason for preferring the Weibull distribution is that in our experience it has proven generally better behaved than the other positively skewed distributions (like the lognormal). The Weibull and the other distributions generally agree in terms of their estimates of median WTP, but may produce very different figures for mean WTP.

With WTP, experience suggests that mean WTP tends to be two or even three times as large as median WTP. We regard median WTP as a conservative, but robust and more reliable, estimate. For this reason, we report median WTP figures for the 5-in-1000-risk reduction in Table 7.4.

Table 7.4 Median WTP for the 5-in-1000-risk reduction beginning now. Wave 1, double-bounded Weibull model. Unclean samples. Annual WTP**.**

	UK	Italy	France*
Median WTP in local currency (s.e. in parentheses)	241 GBP (23)	724 EUR (86)	3144 FF (494)
Median WTP after conversion to € 2002 (s.e. in parentheses)	386 (37)	724 (86)	479 (75)

* We used both wave 1 and wave 2 observations for the France study because of the small sample size.

The VPF implied by these figures is €772,000 for the UK, €1,448,000 for Italy, and €958,520 for France.

Pooled-data models and internal validity tests

To check internal validity, we relate WTP to covariates using an accelerated life Weibull model. Specifically, we allow the scale parameter to vary across individuals, depending on a set of variables thought to be associated with willingness to pay: $\sigma_i = \exp(\mathbf{x}_i\boldsymbol{\beta})$, where x_i is a 1×p vector of regressors, and β is a p×1 vectors of coefficients. In other words, $\log WTP = \mathbf{x}_i\boldsymbol{\beta} + \varepsilon_i$, where ε follows the type I extreme value distribution with scale θ.

We pool the data from the three European countries to increase the sample size and to be able to provide recommendations for VPF figures to use for EC policy purposes. The first specification of this econometric model includes an intercept and an income covariate. The income variable is included in an effort to answer the question whether WTP for the 5-in-1000-risk reduction and the VPF should be allowed to be dependent on a country's income. Other specification includes country dummy variables in order to test whether there are country-specific factors that are influencing WTP additional to the other explanatory variables. Finally, we include age dummies, gender, education, and measures of the health status of the respondent. This specification allows us to check whether the VPF should be adjusted for the beneficiary's age and health status in environmental policy applications. It should be noted that the sign of the age and health status variables is not known *a priori*. One would expect WTP to increase with baseline risk, but higher baseline risk implies lower remaining life, an offsetting effect if the value of each remaining life year is assumed to be constant. Under restrictive assumptions, Shepherd and Zeckhauser (1982) obtain an inverted-U shaped relationship between WTP and age. Similar considerations hold for the health status dummies. One would expect, however, income to be positively correlated with WTP. The sign of education is not known *a priori*: someone with better understanding could give a lower or a higher WTP. The reader is referred to Alberini *et al.* (2004) for details of the econometric tests and results. We summarise the main results:

- The results imply that mean WTP for the 5-in-1000 risk reduction from the three European countries is €1,129 per year (s.e. €132.5), while median WTP per year is pegged at €526 (s.e. €39.5). The implied VPFs are €2.258 million and €1.052, respectively.
- Income is significantly associated with WTP, a result that is consistent with expectations.
- Holding household income the same, the French and the Italian respondents hold WTP values that are greater than their UK counterparts.

- WTP declines only for the oldest respondents in the sample, who hold WTP amounts that are approximately 20% lower than those of the other respondents, all else the same. However, the coefficient on the dummy for a respondent who is 70 or older is not significant at the conventional levels. Still, it is interesting that these results confirm those of the earlier Canada and USA studies (Krupnick *et al*., 2001; Alberini *et al*., 2001).
- As in earlier studies, males have slightly lower WTP and so do people with higher levels of education.
- Persons who have been hospitalised for cardiovascular or respiratory illnesses over the last 5 years hold WTP amounts that are over twice as large as those of all others. The presence of cancer and chronic illnesses, however, does not influence WTP.

Recommended values

Interpretation for VOLY

The discussion of the appropriate WTP metric for the air pollution context above concluded that the epidemiological evidence dictated that the VOLY be adopted. Since we do not have direct estimates of VOLY – our survey generates VPFs – we rely upon a conversion relationship between changes in probabilities of death and changes to life expectancy. This relationship is established in Rabl (2002), which presents the equivalent change in life expectancy associated with the 5-in-1000 change in risk of premature death for different ages and sex, based on EU population statistics. It suggests, for example, that a person of age 55 will gain an equivalent of 40 days from a 5-in-1000 change in risk.

Recommended values for premature death in ExternE (NewExt)

The central values are based on the 5-in-1000 immediate risk change results. Based on the pooled parametric analysis of the data from the three countries (UK, France and Italy) we recommend the value of €1.052 million as a central Value of a Statistical Life (VPF), which could sensibly be rounded to €1 million. We use median values because the econometric analysis suggests that, whilst median values from various assumed distributions agree, the same does not hold for mean WTP. We regard median WTP as a conservative, but robust and more reliable, estimate. A Weibull distribution is taken as it has the best fit out of the alternative distributions (the mean value is €2.258 million).

For use to value air pollution impacts within ExternE we need to convert the WTP for 5-in-1000 immediate risk change into a value of life year lost. Rabl (2002) derives the changes in remaining life expectancy associated with the 5-in-1000-risk change over the next 10 years valued in this study, based on empirical life-tables[10]. According to

[10] A change in the probability of surviving the next 10 years changes the probabilities of surviving all

Rabl's calculations, the extension in life expectancy ranges from 0.64 to 2.02 months, depending on the person's age and gender, and averages 1.23 months (37 days) for our sample. To find out the value of a life-expectancy extension of a month, we divide a respondent's WTP by that respondent's life expectancy extension. A Weibull double-bounded model pegs mean WTP at €1,052 (s.e. 128.4) per year for each month of additional life expectancy. Median WTP is €465 (s.e. 33.3) for a month of life expectancy gain. Because in our survey the payments would be made every year for ten years, the total WTP figures for a life expectancy gain of one month are €10,520 and €4,650 respectively. The implied values of a statistical life-year (VOLY) are €125,250 and €55,800, respectively. Given the uncertainties, this latter number – as a central estimate – might safely be rounded to €50,000.

The VOLY of €50,000 is derived from an annual payment made over a ten-year period and as such does not require further discounting since we assume that the respondents have implicitly done this when giving their answer. Since available empirical evidence suggests that a typical time period of latency to elapse in the case of chronic air pollution-induced mortality is 5-7 years we may adopt this value for chronic mortality impacts, whilst noting that the life years lost (gained) after the time of death are not accounted for in this unit value. If, however, we assume that the VOLY of €50,000 is equivalent to the VOLY derived from life-table analysis, (following Hurley and Miller, 2004; and Friedrich and Bickel, 2001), discounted at 3%, then the equivalent undiscounted VOLY is (50,000/0.67) = €74,627[11]. For calculating new results, this value is rounded to €75,000. This can be interpreted as a value for acute mortality as long as it is assumed that no other factors (e.g. a victim's health condition at time of death) affect WTP for these end-points.

Upper and lower bounds are estimated in the following way:

- The upper-bound value is taken as that resulting from the results from the 1-in-1000-immediate-risk change. The results for this risk change give higher VPF/VOLYs because of respondents insensitivity to scope i.e. the 5 in 1000 risk change is not valued five times higher than the 1 in 1000 risk reduction. We do not have pooled data for this risk change but instead use the UK results. These give a VPF of €3,310,000 and a VOLY (discounted) of €151,110. The corresponding undiscounted VOLY amounts to €225,000 (rounded).
- The lower-bound estimate is derived from the results of the French questionnaire that uses a direct estimate of an equivalent change of life expectancy of €200. This

future periods, conditional on being alive today. The sum of these future probabilities of surviving is a person's remaining lifetime. Rabl's calculations are based on an exponential hazard function, $h(t)=\alpha \cdot \exp(\beta t)$, where t is current age, and α and β are equal to $5.09 \cdot E\text{-}5$ and 0.093 for European Union males, respectively, and 1.72E-5 and 0.101, respectively, for European Union females.

[11] Note that under this approach a zero discount rate would result in acute and chronic VOLYs being the same.

converts to a VOLY of €18,250. The corresponding undiscounted VOLY amounts to €27,240.

- The upper and lower bounds are considerably less robust than the central values because they are based upon survey results themselves derived from much smaller sample sizes (322 and 50 respectively).

Remarks

The preceding sections have outlined how the NewExt project (ExternE, 2004) has made progress in the valuation of premature death resulting from air pollution. It was demonstrated that, whilst the context of air pollution might suggest that direct transfers of other contexts is not appropriate, this is the only procedure possible given the lack of valuation studies in this context. It was also highlighted that the epidemiological evidence suggests that the appropriate metric is the value of life expectancy lost rather than the value of statistical life, on which almost all empirical valuation studies focus.

In order to fill this gap the project team committed to undertake a contingent valuation study in three European countries – France, UK and Italy. The only developed survey instrument designed specifically to address the valuation of death in the air pollution context was that of Alan Krupnick and colleagues from Resources For the Future (RFF) in the USA, and as a sub-contractor to the project team, the project was able to adopt this same survey instrument. As well as benefiting from the RFF's experience of administering the survey in North America, the project significantly reduced the development costs associated with the construction of such an instrument. Nevertheless, the country teams conducted a series of focus groups and/or one-to-one testing in order to better understand how the respondents interpret the questionnaire.

The focus groups, verbal protocols and debriefing have identified possible limitations of the questionnaire:

- Respondents find it difficult to understand small risk reductions and to distinguish risks of 1/1000 and 5/1000;
- Finding it difficult to construct their WTP, the respondents may anchor their response to the starting bid;
- Respondents may doubt the efficacy of a treatment that they have to pay themselves because it is not recognised for reimbursement by the social security system common in Europe, in particular France (the questionnaire had been developed for the USA where the health insurance system is totally different).

In view of these weaknesses the French team tested several variants of the questionnaire (on samples of about 50 each) to explore how it could be improved; in particular a variant phrased in terms of life expectancy gain with open-ended questions.

The project team finds that the recommended VPF and VOLY values are comparable to the central value used by DG Environment, and provide a much-needed empirical validation for current practice in policy analysis. The testing by the country teams does, however, provide some evidence for the argument that we cannot regard these results as the last word on this subject. The three elements of the survey instrument that have been most challenging are outlined in the paragraphs below.

Even given the pictorial representation of the risk changes in the survey instrument and the reinforcing voice-overs, there was some evidence that the small size of the risk changes involved still proved to be difficult for the respondent to be able to provide meaningful values. The scoping tests showed that, although the values for the smaller risk change are lower than the larger risk change, they are not proportional as one might expect.

Some work was undertaken in the French variants of the survey instrument to address this problem by substituting the risk change for the equivalent length of life expectancy, although some respondents questioned the quality of life during the relatively short life extension (of approximately one month). The issue of the appropriate metric, however, remains outstanding for valuing premature death in the air pollution context since the epidemiology seems to dictate the use of values for the change in life expectancy and more future effort in valuing this directly in Europe is clearly required.

There remains a question mark over the effectiveness of using an abstract commodity to be valued. On the one hand it is recognised by Krupnick *et al.* (2000) – and is demonstrated by the French variants – that supplying a public good context is likely to attract a number of biases relating to free-rider effects or altruistic motives. On the other hand, in the absence of a recognisable or familiar commodity there is a tendency to think of health products or services for which individuals have been shown to have different preferences (biased in relation to the real context with which we are concerned).

It remains to be seen whether there is robust evidence of starting point bias being introduced by the use of dichotomous choice in the survey instrument. Preliminary analysis presented in the French report suggests that this might be the case. It is, however, an issue that requires further testing in the European context.

These issues, together with the fact that we would like to establish values on the basis of a larger sample sise, suggest the need for further research in establishing unit values for air pollution-related deaths in the ExternE context. Nevertheless, the values that we derive in this report represent significant progress in this quest and can be regarded as among the most appropriate available at the present time.

7.2.2 Morbidity[12]

In reviewing the morbidity health end-points we use as our starting point the values derived in the recent ExternE work. There has been one major new empirical study on the valuation of these end-points, covering five countries across Europe (Ready *et al.*, 2004) and the pooled results of this study are used in the first instance when discussing the health end-points below.

The starting point for the valuation of health end-points is the identification of the components that comprise changes in welfare. These components should be summed to give the total welfare change, assuming no overlap between categories. The three components include:

- *Resource costs* i.e. medical costs paid by the health service in a given country or covered by insurance, and any other personal out-of-pocket expenses made by the individual (or family).
- *Opportunity costs* i.e. the cost in terms of lost productivity (work time loss (or performing at less than full capacity)) and the opportunity cost of leisure (leisure time loss) including non-paid work.
- *Disutility* i.e. other social and economic costs including any restrictions on or reduced enjoyment of desired leisure activities, discomfort or inconvenience (pain or suffering), anxiety about the future, and concern and inconvenience to family members and others.

The welfare changes represented by components (i) and (ii) can be approximated using market prices that exist for these items. This measure – in best practice – needs to be added to a measure of the affected individual's loss of utility, reflected in a valuation of the willingness-to-pay/accept (WTP/WTA), to avoid/compensate for the loss of welfare associated with the illness.

Note that there is the possibility of overlap between components since, for example, the individual will include both financial and non-financial concerns in his/her assessment of loss of welfare. Financial costs are often not borne fully by the individual but are shared through health insurance and public health care provision. Thus, we assume here that the financial costs are separable and measured in component (i). If this is not the case, then a part of the disutility measured in the WTP estimate will be incorporated in the private medical costs associated with treatment (or prevention) of the health end-point, and the total valuation should be reduced by an equivalent amount.

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[12] Presentation based on CAFE (2005).

Health care resource costs
The generic unit costs for hospital-based health care are presented in Table 7.5. The data have been derived from Netten and Curtis (2000), and MEDTAP International, reported in Ready *et al*. (2004). Since these data are based on public health care provision they are exempted from indirect taxes and are therefore expressed at factor cost. It has not been possible to derive unit cost data for all EU countries, but mean values calculated from the available data are presented and can be used as a first proxy for EU countries that currently do not report such values. Generic hospital costs are the average costs of a wide variety of specialist treatments, for use when precise information about the nature of the individual's hospital contact is not known.

The outpatient value for the UK is significantly higher than those in the other countries listed. This suggests that a different cost definition may have been used in its derivation – although this has not yet been established. The mean value, excluding the UK value, is €23, compared to the value of €33 when the UK figure is included. We suggest, for the present, that the higher value should be used as the central value, with the lower figure used for sensitivity analysis.

For cardiology, the inpatient unit cost is 1.92 higher than the generic unit cost. This multiplier may then be applied when heart-related conditions are considered, in the discussion of end-points below.

Table 7.5 Generic unit hospital health care costs (€ 2000 prices).

Country	Emergency room/outpatient: cost/visit	Hospitalisation: cost/inpatient day
Belgium	19	241
France	29	375
Germany	24	321
Italy	20	256
Netherlands	30	390
Spain	27	345
UK	96	330
Mean (EU)	35	323

Source: Netten and Curtis (2000), Ready *et al*. (2004)

Other unit cost data for more minor health conditions are presented in the discussion of the individual health end-points below.

Costs of absenteeism
The costs of absenteeism adopted in this study are based on figures contained in Confederation of British Industry (CBI, 1998). This report is the outcome of a survey on absence conducted by the CBI. The survey aims to provide a comprehensive guide

to levels, causes and costs of absence in the UK. Respondents to the survey were asked to quantify the direct cost of absence. The direct cost of absence is based on the salary costs of absent individuals, replacement costs (i.e. the employment of temporary staff or additional overtime) and lost service or production time. The survey included a wide range of organisations – 45% from manufacturing, 34% from services, 19% from the public sector and 2% from other types of organisation.

The mean direct cost to business per employee-day absence is €114. However, the mean cost estimates are skewed (increased by the fact that a small number of employers have very high costs). From consideration of the structure of the survey the authors concluded that the median estimate was likely to be a better indicator of average costs (CBI, 1998, p13). Based on the median, the average cost per employee is lower at €85. It should be noted, as an aside, that by using these (direct) unit cost estimates, it is implicitly assumed that the wage rate will remain unchanged, even with no absenteeism.

Respondents to the survey were also asked to provide an estimate of the indirect costs of absence. Indirect costs relate to lower customer satisfaction and poorer quality of products or services leading to a loss of future business. The indirect cost/day is estimated at €168, although there is less confidence in this value because of a relatively low survey response rate for the question from which the value is derived. Its representativeness is therefore not fully established.

The figure for indirect costs should be added to the direct cost estimate to obtain the total cost of absence per employee of €253/day. However, given the lower confidence we have in the indirect cost estimate, it may be preferable to use the combined figure for sensitivity analysis, with the median direct cost estimate of €85/day as a central estimate. A crude alternative is to use the information given in the EUROSTAT Statistical Yearbook on mean annual gross earnings paid to EU employees and divide this by data on the size of the labour force to give a value of marginal productivity – assuming wages equal marginal productivity. This gives a value of €56. However, this estimate does not include all costs (direct or indirect) associated with absenteeism and should therefore only used as a lower-bound estimate for this component.

In order to derive country-specific estimates of the direct and indirect costs presented for the UK by the CBI, we suggest scaling the EUROSTAT country data relative to the EUROSTAT data for the UK and applying these scaling factors to the values derived from the CBI study. Where the data are not available, we use the country purchasing power parity relative to the UK to derive appropriate scaling factors. Mean values across the EU are €58, €88 and €261 for low, central and high values respectively. In aggregating the costs below, we use the central value of €88.

Hospital admissions (HA)

Respiratory hospital admissions are one of the most widely studied health end-points, in Europe and internationally. Their quantification raises important questions about pollution mixtures and background rates of hospital usage. Results from other studies suggest however that the monetary value of their impacts is not high, compared with mortality from long-term exposure.

Ready *et al*. (2004) have estimated a WTP for respiratory hospital admissions in a survey-based approach (contingent valuation method) where the patient stays in hospital receiving treatment for three days, followed by five days at home in bed. The mean value is given as €468 per occurrence. In addition there will be productivity loss for 8 days of €704 and costs of hospitalisation for three days at €969. This gives a total economic estimate of €2,141 per Hospital Admission from respiratory distress. Adjusted from price year 2003 to 2000 this gives a figure of €2,000. This estimate is very similar to that derived by Otterström *et al.* (1998) for a general HA episode, independent of whether this is for a respiratory, congestive heart failure, ischaemic heart disease or cerebrovascular HA. We therefore adopt this common value for these end-points.

Emergency-room visits for respiratory illness

The Ready *et al*. (2004) study derives a WTP valuation for this health end-point over and above the hospital costs. It is described as a visit to a hospital casualty department, required for oxygen and medicines to assist breathing, followed by five days at home in bed. The mean unit value in the five-country pooled study is €242. To this estimate one should add the estimated productivity loss for five days in bed, which is €440. The health service costs of an emergency-room visit should also be added (i.e. €35). Thus, the economic value of an ERV is €717 (2003 prices) or €670 (2000 prices).

Visit to a doctor: asthma and lower respiratory symptoms

Ready *et al*. (2004) found a WTP to avoid a day of asthma attack (excluding medical care and lost productivity costs) of €67, €139 and €295 per day for adult non-asthmatics, adult asthmatics and asthma attack among the respondents' own children, respectively. These were the values for a sample of respondents who were asked to express their WTP to avoid one additional day of asthma attack (in addition to what they had experienced in the last 12 months). The corresponding asthma daily values for a sample that was asked to value an additional day to 14 days were €14, €15 and €42 respectively. The study suggests using the marginal day value of €15 as a central unit value.

Netten and Curtis (2000) give unit values for the resource costs of the general practitioner (GP) in the UK. Here, we use these as representative for typical EU costs. These vary between €25 and €42 depending on whether the consultation period is 9.36 minutes or 12.6 minutes (the two unit periods suggested) and whether qualification costs are included. We assume the longer period to be more realistic. A value of €42 should therefore be added to the WTP values identified in the previous paragraph.

For lower respiratory symptoms a value of €38 may be used. This value was derived for the symptom described as "a persistent phlegm cough occurring every half-hour or so, and lasting one day". GP costs of €42 should be added, giving a total of €80 (2003 prices) or €75 (2000 prices).

Note the end-point here is asthma-related visits to a doctor – not new cases of asthma. The latter have higher costs. In this context, there has been recent work in the UK (http://www.hse.gov.uk/ria/chemical/asthma.htm) that estimated the cost of a new case of asthma at between £42,000 and £45,000. These costs include: loss of income through absence from work or having to change jobs; medical treatment; and pain and suffering. In €, these costs would be broadly €60,000 per new case.

Restricted activity days (RAD)
A value of €148 is available from the Ready *et al*. (2004) study. Here, the symptom is described as three days confined to bed, where there is shortness of breath on slight exertion. Since this value is to avoid an episode lasting three days, the estimate has to be divided by three. To this may be added the EU average per diem productivity loss, dependent on the severity. Thus, one RAD can be valued at €49 or €137 (2003 prices, equivalent to €46 or €130 in 2000 prices) depending on how the end-point is defined (i.e. €49 + €88)[13].

Respiratory symptoms in adults and children with asthma
The asthma attack values given above for adult asthmatics – €139 per event and €15 per extra day – may be used. For asthma attacks among the respondents' own children the WTP per event was €295, and a WTP of €31 for each additional day of asthma symptoms. The value of €38 used for lower respiratory symptoms may be used instead but it is judged that the asthma value, whilst not the end-point being valued, allows us to consider the WTP values of people who suffer regularly from a similar condition. All these values are derived from the Ready *et al*. (2004) study.

Respiratory medication use by children and adults
Regular use of respiratory medication includes the use of bronchodilators. The resource costs of drugs typically associated with bronchodilators vary between €0.5 and €1 per day, according to whether Terbutaline or Albuterol is used[14]. We do not have any evidence for the value of disutility of using bronchodilators and so factor this in implicitly by assuming the total unit value for these end-points is at the upper end of the range presented above, i.e. €1 per day. We do not differentiate between children and adults since use rates of bronchodilators – and therefore unit costs – are assumed to be the same for both groups.

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[13] Note there may be an issue here with the application of this value in the context of the original study (consistency).
[14] http://www.fpnotebook.com/LUN118.htm

Chronic bronchitis (new cases)
There are questions about the approach that has been used by ExternE for the quantification of chronic bronchitis (CB). It is important to resolve them because this end-point has the second largest contribution to the total damage cost, due to the high unit cost that has been assumed. The first question concerns the wide range of severity of different cases of CB. The CRF (concentration-response function) has been based on the study of Abbey *et al.* (1995), but the symptoms in this study are very light (persistent cough or phlegm during at least two months) compared to the severity levels implicit in the only available monetary valuation studies (Viscusi, Magat and Huber, 1991; Krupnick and Cropper, 1992). While some cases are mild and temporary, CB can be a truly debilitating permanent condition, making it impossible to work or lead a normal life. The monetary valuation of Viscusi *et al.* was based on severe cases, with a questionnaire that was applied to the general population. Krupnick and Cropper 1992 used a slightly modified version of the questionnaire of Viscusi *et al.,* but in contrast they applied the questionnaire only to individuals who knew someone with CB. Assuming that their sample was representative, the results of Krupnick and Cropper thus implicitly assume the average distribution of severity levels.

The assumptions about severity levels must be consistent between CRF, background rates and monetary valuation. As for the CRF, the study of Abbey *et al.* yields the RR (relative risk) for an increase in CB due to an increase in ambient concentration. Looking at RR results of a large number of epidemiological studies, one finds that the RR per concentration is fairly similar across a wide variety of morbidity end-points. This suggests that the RR of Abbey *et al.* is likely to be appropriate even for other severity levels. Data for incidence rates are presumably for the average distribution of severity levels. With these plausible assumptions the CRF, background rates and monetary valuation are thus consistent if the latter is based on Krupnick and Cropper. The values found by these authors seem more realistic than those of Viscusi *et al.* because someone familiar with CB is better qualified to indicate a willingness-to-pay (WTP) to avoid the condition than someone who lacks this experience.

One difficulty in applying the paper of Krupnick and Cropper is that their primary purpose was the development of the valuation methodology rather than the provision of numbers that could be used for policy. Their tables contain many different unit costs, for the two variants of the questionnaire that the authors tested and the trades (risk-risk or risk-income) offered for the WTP solicitation. The numbers in the tables range from \$0.53 million to \$1.6 million for the medians. But the only value explicitly mentioned in their text is \$0.4 million; it is based on the risk-risk trade where the risk of CB is traded against the risk of dying, combined with a VPF of \$ 2 million (chosen by the authors to convert the risk tradeoff to monetary values). If one takes the ratio of these values for CB and VPF, together with the new VPF of ExternE (2004) of €1.0 million, one obtains the unit cost of CB as €0.2 million. No adjustment for inflation or exchange rate is needed because the costs of Krupnick and Cropper are used only as ratio.

There are other possibilities for extracting a unit cost from Krupnick & Cropper and/or Viscusi *et al.,* for example the method used by USEPA (Abt Associates, 2000 and 2004) who obtain a WTP to avoid CB of $0.33 million. But that necessitates an assumption about the frequency distribution of severity levels and adjustments for inflation (and, for the transfer to Europe, the exchange rate). For ExternE we recommend the value of €0.2 million because we find its derivation better justified and more transparent. It is very close to what ExternE has used in the past (€0.17 million in 2000).

Other end-points

The Ready *et al*. (2004) study also notes that one cough day is estimated to be €41/day (2003 prices, €38/day in year 2000 prices). The same value should be applied to minor RAD (restricted activity day) and symptom day (note that this is probably a low estimate for a symptom day as one day with mildly, red watering, itchy eyes and runny nose is valued at €53.5). A work loss day is valued according to the discussion of the costs of absenteeism, above. Hence a central value is €88, with lower and upper bounds being €58 and €261 respectively. Translating from 2003 prices to 2000 gives a central value of €82 in a range of €54 to €245 per work loss day.

Summary of health end-points

Table 7.6 summarises the values that have been used in the current study.

Table 7.6 Summary of morbidity values.

Health end-point	Recommended central unit values, € price year 2000
Hospital admissions	2,000/admission
Emergency Room Visit for respiratory illness	670/visit
General Practicioner visits: Asthma Lower respiratory symptoms	 53/consultation 75/consultation
Respiratory symptoms in asthmatics: Adults Children	 130/event 280/event
Respiratory medication use – adults and children	1/day
Restricted activity days	130/day
Cough day	38/day
Symptom day	38/day
Work loss day	82/day
Minor restricted activity day	38/day
Chronic bronchitis	190,000/case

7.3 Valuation of Amenity Losses from Noise[15]

Environmental valuation methods, both stated preference (SP) and revealed preference (RP) methods, have been employed to estimate the economic value of changes in noise levels. Most studies have applied the RP approach of Hedonic Price (HP) to the housing market to analyse how differences in property prices reflect individuals´ willingness-to-pay (WTP) for lower noise levels. More recently there has been an increased interest in applying SP methods to value noise. Contingent Valuation (CV), Conjoint Analysis (CA) and Choice Experiments (CE) have all been applied to value transportation noise.

In order to establish interim values for noise from different transportation modes (air, road, rail) to be used in cost-benefit analyses (CBAs) performed by the EC, there is a need for an overview and evaluation of the valuation techniques, empirical noise valuation studies and the potential for benefit transfer of noise values

The EC project UNITE developed the impact pathway approach for health effects from transport noise. Hunt (2001) reports the preliminary results from the noise impact valuation work in UNITE. He identifies the following five end-points from exposure-response functions, established by TNO (2001), as starting points for the economic analysis:

- Ischaemic heart disease/myocardial infarction;
- Hypertension;
- Subjective sleep quality (Sleep disturbance);
- Speech interference in offices (communication disturbance);
- Annoyance[16].

However, there seems to be no easy way of isolating the economic estimate of annoyance from sleep and communication disturbance in order to avoid overestimation of benefits of noise-reducing measures when aggregating over end-points of ERFs. Therefore, it can be argued that an economic estimate for annoyance could serve as an indicator of the overall impacts of noise (but most probably providing a lower economic estimate of noise impacts).

A review of the values used for noise in four European countries (UK, Denmark, Sweden and Norway) shows that the methodological approach and unit used to measure the economic value of noise annoyance differ between countries, and even

[15] This review of noise valuation is based on Navrud (2002)
[16] Annoyance is defined by TNO (2001), as a feeling of resentment, displeasure, discomfort, dissatisfaction, or offence when noise interferes with someone's thoughts, feelings or actual activities.

between different sectors/agencies in the same country. However, there seem to be two main approaches:

i) An economic value per decibel per year; measured by the Noise Depreciation Sensitivity Index (NDSI), defined as the average percentage change in property prices per decibel.
ii) An economic value per year per person (or household) annoyed by noise. Two measures are used. a) value per person "highly annoyed", and b) value per person "annoyed", independent of the level of annoyance.[17]

The first approach is based on domestic Hedonic Price (HP) studies and/or a review of HP studies internationally; and in a few cases also expert assessments by real estate agents have been used. Nearly all of these studies report the results in terms of the Noise Depreciation Sensitivity Index (NDSI), which gives the average percentage change in property prices per decibel. To convert this capitalised value of expected future rents into an annual value, we have to make assumptions about time horison and discount rate (which also vary between countries). To avoid making these assumptions, several authors (e.g. Palmquist, 1981) have suggested using rental charges instead of sales prices as the dependent variable in HP regressions. Soguel (1991, 1994) used the monthly rent (net of charges) as the dependent variable in his HP regression on dwellings in the town of Neuchatel in Switzerland. He found a value of SF 5.85 per dB per household per month, which equals about €47 per year per household (1 SF = €0.675). Furlan (1996) and Locatelli Biey (1994) also used monthly rent of apartments in their HP studies in the inner city of Paris and Turin (Italy), respectively. The last study used traffic volume as a proxy for the noise level, while Furlan *op cit* had noise level data. However, neither of these two studies collected data on the income of households, and do not contain data on the average market price of apartments. Thus, no estimates of WTP per household can be constructed. One problem in using rental charges in HP studies is that the rental market could be controlled and therefore the difference in noise level often would not be fully reflected in differences in rental charges.

The second approach is based on Contingent Valuation (CV) and Choice Experiments (CE) like Conjoint Analysis (CA), and most of these valuation studies have been conducted over the last 5 – 10 years.

In addition to these two approaches, there have also been studies that try to calculate the national costs of noise annoyance in terms of percentage of Gross Domestic Product (GDP). However, these results are not very relevant for benefit transfer to CBAs of noise-reducing measures.

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[17] Sometimes values are also expressed as per person *exposed* to noise levels above a certain level e.g. 55 dB without referring to any annoyance level. This means that persons exposed to, but not annoyed by, noise will be included

The recommended economic values for noise annoyance vary. This could be due to different initial noise levels, different income level, cultural differences, different methodological approaches (and the noise valuation unit used), whether other social costs than the annoyance costs are included, etc. In the sub-sections below we summarise the results of studies relating to different transport modes.

7.3.1 Road traffic noise

NSDIs for road traffic noise have been reported ranging from 0.08 % to 2.22 %, see Bateman *et al.* (2000). Bateman *et al.* (2000) conclude that noise researchers have suggested that an "average" value lies somewhere in the lower part of this range. A simple mean of these studies suggests a NSDI of about 0.55. A HP study, not included in this review, using rental charges for apartments in Paris (exposed to road traffic noise levels between 50 and 80 dB(A)) should also be mentioned. Furlan (1996) found a NSDI of 0.20 – 0.33 %.

Bateman *et al.* (2000), in their review of studies, point out that the use of a single statistic to compare studies conceals considerable heterogeneity in the exact method of their application. As an example, each of the studies deals with noise in a slightly different manner. Whilst the majority of studies have used the L_{eq} measure of noise, the method by which the noise pollution impacting on a particular house is assessed can be very different from study to study. A number of studies adopt the noise contour approach whereby data from various monitoring points are used to construct bands of similar noise pollution across the urban environment. The noise pollution experienced by any particular property will depend on the band in which it falls. Studies using this approach include Gamble *et al*. (1974). More advanced measures of noise pollution can be achieved by using models that take account of the exact characteristics of a particular dwelling. Data from these models are likely to be much more accurate. Studies taking this approach include Pommerehne (1988), Soguel (1991) and Vainio (1995, 2001). Bateman *et al.* (2000) also observe that studies vary considerably in the choice and accuracy of the explanatory variables used in the regression analysis and in the choice of functional form, and this affects the level of the observed NSDI.

Results from SP studies on road traffic noise show the wide range of values per dB per household per year. If older SP studies (done before 1995 and using exposure-based scenarios) are excluded and only studies included, which value *reductions* in noise levels a range of €1-27 per dB per household per year can be observed. This range of values reflects a combination of differences in methodological and modelling approaches, and differences in preferences, sites, institutions, culture and contexts. A meta-analysis of these studies, which could test the significance of these explanatory factors, could not be carried out, because there were too few studies to perform comprehensive meta-analysis.

7.3.2 Aircraft noise

Gillen and Levesque (1989) in their review of 15 HP studies on aircraft noise (and one combined HP and Expert assessment) in mainly US cities found NDSI in the range from 0.4 to 1.1 % per dB, with a median value of 0.5-0.6 %. Another review, including also recent HP studies, Bateman *et al.* (2000) found reported NSDIs (i.e. the percentage decrease in housing prices following a 1 dB increase in noise pollution) in the range from 0.29% to 2.3% for aircraft noise. The variety of NSDI values should not come as any surprise. Theoretically, we would not expect different housing markets to have the same hedonic price function and, therefore, would not expect applications of the hedonic pricing technique in different cities in different years to return identical results.

7.3.3 Rail noise

Only two original valuation studies on rail noise have been identified; both of them HP studies. However, the CV scenario, annoyance level questions and noise exposure data of Navrud (2000) also include railway noise.

Strand and Vågnes (2001) used both HP and Delphi studies of real estate brokers in one part of Oslo (Gamlebyen near the main railway station (using a Multi Criteria Analysis technique). Using distance to the rails as a proxy of noise, for semi-detached and single family houses this HP study finds that a doubling of the distance to the tracks would mean a 10 % increase in property prices. In the Delphi study a mean WTP of 2,000 1996 NOK per metre increased distance to the track; all results are for apartments. For single family and detached houses the impact is 20-27 % higher than for apartments.

A HP study on railway noise in Sydney, Australia (Holsman and Paparoulas, 1982) found that the occurrence of railway noise in areas with no benefits from increased accessibility reduces property prices by 10 %.

7.3.4 Industrial noise and other types of noise

No valuation studies specifically on industrial noise have been identified. However, the HP study of Oosterhuis & Van der Pligts (1985) looked at both road traffic noise and industrial noise. They found a NSDI of 0.4 % for the combined impact of the two noise sources. The CV scenario, annoyance level questions and noise exposure data of Navrud (2000) also included rifle range noise and industrial noise (but no noise exposure data for the latter).

7.3.5 The potential for benefit transfer of existing studies

The noise valuation literature is dominated by HP studies (most of them old) on road traffic and aircraft noise of varying quality. However, NDSI estimates from HP studies seem to be problematic to transfer, both theoretically and in practice (Day, 2001).

There is an increasing number of SP studies on road traffic noise, but only a few present WTP in terms of "€ per annoyed person per year" for different annoyance levels, which correspond to end-points of ERFs. Due to the low number of studies that can be used for this approach, a "second-best" alternative is to evaluate all these SP studies with regards to quality (e.g. avoid using studies with scenarios based on changes in exposure rather than annoyance and health impacts), choose the best ones, and calculate a value in terms of "€ per dB per person per year". The number of high quality European studies on road traffic noise might be sufficient to establish an EU value based on this approach. For noise from air, rail and industry there seem to be too few SP studies to evaluate whether the same values as for road traffic noise can be used. Due to the different characteristics of these four types of noise, one would expect that these exposure-based values would differ between different noise sources (while the preferred annoyance-based unit value would probably not be so sensitive to the source of noise). Another uncertainty the per dB approach faces is the conversion of WTP values for relatively large discrete changes in noise valued in SP studies to marginal values assuming linearity. Benefit function transfer might be used to reduce this uncertainty.

In addition to benefit transfer in space, one might also have to transfer values in time. This is usually one using the consumer price index (CPI) as a proxy. However, it is still an open question whether the CPI of the study country or the policy country should be used. Also, one should consider whether the CPI is representative of the change in value over time for noise annoyance.

7.3.6 What should be the cut-off point for valuing noise ?

When using economic values per dB, the practice among transportation authorities in Europe and the USA has been to use different cut-off points for different modes of transportation. Typically a "bonus" of 5 dB is given to rail, compared to road and air to correct for the fact that rail noise at the same noise level is less disturbing than road traffic and aircraft noise. This means cut-off points of 55 dB for air and road, and 60 dB for rail, which means zero damage costs of noise below these levels.

Exposure-response functions for transportation noise show that people are annoyed by noise at levels below 55 dB (Miedema and Vos, 1998, 1999 and Finegold *et al.,* 1994), and that elimination of noise annoyance occurs at 37-40 dB (and theoretically even lower, but in practice other noise sources, e.g. noise from neighbours, would dominate at lower levels of transportation noise). The review of valuation studies also shows that people exposed to noise levels below 55 dB and/or not annoyed by noise have a positive willingness-to-pay (WTP) for noise-reducing measures like noise-absorbing road covers and improved tyres (see e.g. Navrud, 1997). To avoid underestimation of the benefits of such measures, which reduce road traffic noise for both high and low levels of initial noise (as opposed to noise screens in locations with high noise levels for example), the cut-off point for noise should be below 55 dB. However, both ERFs and economic value estimates for annoyance become very uncertain below 50 dB due to the few empirical studies at these low noise levels. Thus, L_{den} 50 could be used as an interim cut-off point for economic valuation. However, even this cut-off point will most probably produce conservative estimates (underestimates) of benefits from reduced noise annoyance, which could lead to "wrong outcomes" of CBAs of noise-reducing measures which also have a positive impact on a high number of houses with low initial levels of noise.

7.3.7 Same value for noise from different sources?

The noise measure L_{den} corrects for different distribution of noise over time, but not the content and composition of noise. An example: a classical music concert and a rock music concert might have the same L_{den} level, but the noise has very different content and composition, and the enjoyment/annoyance of these two concerts would vary among individuals according to their preferences (an important difference between this example and transportation noise is that, in most cases, individuals are involuntarily subjected to the latter noise source).

Aircraft noise is often considered to be the worst since it is characterised by infrequent events with very high noise levels. Rail noise has the same characteristics, but in contrast to aircraft noise you can hear the train coming well in advance and prepare for the high noise level when it is passing. Also it is easier to find effective noise-reducing measures against rail noise, while it is more difficult to protect households from air noise (i.e. noise coming through the roof). However, if there are few or no restrictions on night traffic, train noise causes high levels of sleep disturbance. In situations with restrictions on rail noise during the night, road traffic noise is ranked higher in terms of noise annoyance than rail, but lower than air. Road traffic is characterised by more frequent and constant levels of noise than air and rail noise. The annoyance from industrial noise will vary depending on the type of industry and noise. Single tone component noise is more disturbing than noise over a wide spectrum, and sharp increases in noise levels (e.g. hammering) are more disturbing than a constant noise level (e.g. ventilation system, fans). Thus, the same L_{den} level for different sources gives different levels of annoyance. This is also reflected in ERFs for noise from different sources.

Results from an HP study in Glasgow including data on both aircraft and road traffic noise in Glasgow also indicate that reductions in aircraft noise are valued higher than road traffic noise (Bateman *et al*., 2000; table 9-3).

In a situation where individuals are exposed to *multiple sources of noise,* measures to reduce one dominating source (especially if the decibel level is below 65 dB(A) or one of two equal noise sources will have little effect on the level of annoyance as the other sources will take over and dominate (e.g. shutting down an airport makes people at some distance from the airport more aware of and annoyed by nearby road traffic noise). Therefore, action plans towards noise must consider all noise sources (especially when the noise level is below 65 dB(A); at higher noise levels there is a more significant effect of reducing one noise source, and they may be treated source by source). Also, the effect on total annoyance by different environmental factors might be little affected by a measure to reduce noise from one or several sources if for example levels of air pollution (causing health impacts and visibility effects), visual intrusion and accident risks are constant. Therefore, one should shift the focus from noise alone to look at the total annoyance level and welfare effect of all environmental factors that affect households.

If we use annoyance-level-based units of value, we should be able to use the same value for all noise sources (since the difference between noise sources is "taken care of" in the different ERFs between noise levels and noise annoyance), while noise exposure-based values would have to be different for different noise sources to correct for their different characteristics and level of annoyance at the same dB level.

7.4 Valuation of Impacts on Building Materials, Visibility and Transmission Lines

7.4.1 Cultural and historical heritage

AEAT (2003) argues that, although damage to cultural heritage in Europe has been one of the driving forces behind the early actions to deal with acid rain, progress to quantify these damages in economic terms has been slow. The authors claim that the reason for this involves the uncertainties in the quantification process, mainly the lack of an inventory of the European stock at risk[18]. Another reason is that maintenance costs for historical buildings are likely to be variable, in contrast to the maintenance costs observed for houses, for example, which can be taken from builders or architects. Some

[18] Other uncertainties are: (i) application of a limited number of response functions to materials that will vary in some ways from the experimentally exposed samples; (ii) extrapolation of response data from small samples to materials used on buildings which differ in their exposure characteristics; (iii) determination of the critical thickness for the different materials; (iv) assumption that building owners react to material damage in a purely national manner (AEAT, 2003).

study cases are available for specific monuments in Europe and were reviewed, although extrapolating their results for Europe is acknowledged to be incorrect given the lack of an inventory of stock at risk and other data. These studies are summarised in Table 7.7.

Table 7.7 Review of cultural heritage valuation studies associated with air pollution.

Study and nature of the asset	WTP (€) [a)]	WTP definition [b)]	Annuity[c)]
Pollicino and Maddison (2002) Lincoln Cathedral, UK	1-2 per year of soiling, residents of Lincolnshire	Household, annual, double-bounded dichotomous choice, tax	**1-2**
Morey *et al.* (2002) Monuments, Washington, DC	16 - low impact 23 - medium imp. 33 – high impact	Household, one time only, conjoint analysis, none	1.0 1.5 2.1
Navrud and Strand (2002) Nidaros Cathedral, Norway	51 – originality preserved 45 – restoration losing originality	Individual, annual, open ended, tax and donation	51 45
Grosclaude and Soguel (1994) Historical buildings, Neuchatel	77 - 86	Individual, annual, bidding game, donation	77 - 86

a) Using average exchange rates for the year of the study;
b) Individual or household; periodicity; elicitation format; payment vehicle.
c) € Estimated annuities were calculated for a time horizon of 50 years using a discount rate of 6%.

An alternative methodology for quantification and valuation of soiling effects has been proposed by Rabl (1999), who looked at total soiling costs (the sum of repair cost plus amenity loss) to show that for a typical situation where the damage is repaired by cleaning, the amenity loss was equal to the cleaning cost (for zero discount rate). The total damage costs are twice the cleaning costs. The author argues that the first of these costs (repair costs) is the most straightforward, because cleaning and repair of buildings involve market transactions that are relatively easy to measure. Valuing the loss of amenity, such as the aesthetic loss as a building that becomes dirty, involves subjective perception and might necessitate a specific contingent valuation study.

However, continues the author, this subjective valuation is reflected in market decisions about cleaning and repair, and the amenity loss can be inferred from the cleaning and repair costs. The key assumptions involves (i) the amenity loss is restored by renovation (cleaning and repair); (ii) people minimise total cost; (iii) the decision to clean or repair is made by the people who suffer the amenity loss.

Rabl (1999) recommended the following function:

$S_i = a \cdot P_i \cdot \Delta TSP_i$ (where $a = b \cdot 2$)

S_i = Annual soiling damage at receptor location i.

P_i = Number of people in location i.

ΔTSP_i = Change in annual average TSP (Total Suspended Particles) $\mu g/m^3$.

a = WTP per person per year to avoid soiling damage of $1\mu g/m^3$ particles.

b = Cleaning costs per person per year from a concentration of 1 $\mu g/m^3$ of TSP.

This function allows a site-specific assessment, linking reductions in particle concentrations with population. In applying this function, a number of considerations are important:

- PM_{10} may not be the most relevant functional unit for analysis. Instead black smoke or TSP (total suspended particulates) is a better metrics for assessing damages.
- Knowledge of the characteristics of different types of particulates suggests that only primary particles have soiling effects. AEAT (2003) assumes that secondary particles formed from SO_2 (for example, sulphate aerosol and ammonium sulphate) and from nitrates (for example, ammonium nitrate and nitrate aerosol) are very different in nature (they do not contain PEC – Particulate Element Carbon) and do not lead to a loss of reflectance. For national or city-wide measurement data, the use of measured PM_{10} would therefore need adjustment for the proportion of primary and secondary particulates in the original air pollution mixture. In Rabl (1999), this implied an increase by a factor of three, assuming that $PM_{10\,(primary)} = PM_{10}/3$.
- Regarding the threshold for soiling, the loss of reflectance needed to trigger action (cleaning), will only occur when there is a certain build-up of particles. For this reason, observation shows that soiling is only associated with urban emissions of particles – there is no rural effect from low levels of building exposure. It is even likely that the effect is constrained to certain road types, notably street canyons, where buildings are extremely close to the roadside.

Rabl (1999) argues that the above arguments apply also to historical monuments and buildings, in the sense of collective decision-making: public expenditures for the restoration of historical buildings are a reflection of our collective willingness-to-pay. It is meaningful to look at restoration costs, with the pragmatic purpose of quantifying such costs in order to establish consistent reference values for the efficient allocation of limited financial resources; it is not the purpose to pass judgment on the intrinsic value of the damaged objects. The author takes the expenditures for the renovation of historical buildings as *de facto* expression of society's valuation, and uses it to account for loss of amenity. We await new data for stock-at-risk in order to up-date results presented by Rabl.

7.4.2 Visibility

Analysis in the USA has concluded that reduced visibility is one of the major impacts of air pollution. 'Visibility' here relates to a reduction in visual range through the presence of air pollutants, especially particles and NO_2, in the atmosphere. In Europe, however, the issue has received very little attention. Indeed as far as we are aware no new valuation studies have been undertaken in Europe since the previous ExternE report in 1998. New work has been undertaken in the USA, most notably by ABT Associates (2000).

To estimate the value of the many different visibility improvements that would result in different locations from an implemented air pollution policy or regulation, ABT (2000) proposed to use evidence from these studies to estimate a general relationship between the amount of improvement in visibility and the average value that households place on such an improvement. One plausible relationship that has been fitted to the available recreational visibility values assumes that what matters is the percentage of the change and not the absolute change in visibility. This relationship says that household willingness to pay for a change in visual range from v1 to v2 is a coefficient times the natural logarithm of (v2/v1). The coefficient is estimated using the reported willingness-to-pay estimates and the corresponding (v1, v2) pairs of visual range in studies that value visibility improvements. For example, using willingness-to-pay estimates for visual range improvements in national parks in the Southeast, Chestnut and Dennis (1997) reported estimated coefficients of 85 and 50 (based on 1994$) for in-region and out-of-region households, respectively. Using these fitted relationships, household willingness to pay for any percent improvement in visibility at national parks in the Southeast can be estimated. Using a general relationship based on the information in the recreational visibility valuation studies, the average household value for *any* visibility improvement in a national park can be estimated (ABT, 2000).

Similarly, a general relationship between visibility changes in residential areas and household values for those changes was estimated by ABT (2000), based on residential visibility valuation studies. Assuming the relationship to be the same in all urban areas, the reported information from all the studies can be used to estimate a single relationship. If, alternatively, one allows that the relationship may vary from one urban area to another, then one can estimate the relationship that best fits the information available for each city. Chestnut and Dennis (1997) did the latter, using the same general form of relationship that was used for national parks (in which the percent change in visibility, rather than the absolute change, is what matters). Using these city-specific functions, they calculated what a twenty percent improvement in visibility would be worth in each city. There was a broad range. For example, a twenty percent improvement in residential visibility was worth as little as $22 a year per household (in Cincinnati) and as much as $272 a year per household (in San Francisco). Results for both recreational and residential contexts are presented in Table 7.8 below.

Table 7.8 Annual unit costs per household for visibility improvements.

Visibility improvement (% of visual range)	US$ (1999)		US$ (1997)	
	Recreational		Residential	
	Minimum	Maximum	Minimum	Maximum
10%	7	10	---	---
20%	11	19	24	278
100%	42	69	---	---

Source: Adapted from ABT (2000).

However, given the lack of concern about air pollution in Europe on visibility (AEAT, 2001; AEAT, 2003), using values based on the US experience to evaluate policies regarding visibility effects of air pollution in Europe would be inappropriate due to the uncertainties involved in the benefit transfer. In the absence of a specific contingent valuation study for Europe aiming to elicit the average willingness-to-pay measure to improve visibility, some adjustment in the US numbers may be done to account for this lower concern about visibility effects. It is not clear on what basis this might be done, however, and we suggest that policy measures that are thought to entail discernible effects on visibility be assessed in qualitative terms only, or a new valuation study undertaken to supply more credible numbers.

7.4.3 Transmission lines

The ExternE report *Extension of the accounting framework 1997* identifies the following impacts from transmission lines:

- Impact of electro-magnetic fields (little evidence for);
- Impact on bird populations;
- Land use and forest cutting impacts;
- Visual amenity (most significant);
- Noise;
- Risk of accident (risk of plane etc. – close to normal. Ordinary risk very low);
- Traffic and disturbance during construction.

Of these, the valuation of the visual amenity aspects of power transmission lines appears to be recognised as being potentially the most significant externality. This is so, because economic development demands a larger geographical coverage of electricity provision at the same time as landscapes – and particularly specially conserved natural areas – are valued by the general population.

In previous ExternE work, visual impact valuation studies from other contexts have been used to identify possible externalities from power transmission lines. This has been fraught with the usual difficulties relating to benefit transfer, and exacerbated by the problem that the good to be valued was not common between study site and policy application. One new study (Atkinson *et al*., 2004) provides an improvement on this since it conducts a contingent valuation survey to assess the size of the visual amenity conferred on local landscapes by replacing the overhead electricity transmission towers with those of alternative designs. Survey respondents were asked to rank six tower designs including the current "Lattice" design. Respondents who ranked any new design as being preferable to the current one were asked to express their WTP to see specified towers in their area changed to this new design. Details of the study results are in the full report; however, the main features of the results are that the least favoured of the six designs generated a negative WTP whilst the most favoured design generated a mean WTP of €10 per household.

Clearly this study adds greater richness to the type of analysis that can be undertaken in incorporating the external costs into energy policy design. There remains, however, a gap for studies that measure the WTP for avoiding the imposition of transmission lines at all, in a given landscape.

7.5 Valuation of Crop Losses

We simply note here that the application of the impact-pathway approach relies upon up-to-date data. Thus, in Table 7.9 below we show updated prices of the crops damaged by air pollution. Prices have changed significantly in recent years, those for important crops such as wheat and potato have gone up. This may well completely offset the changes in ER-functions (reported in section 6.7.2) and significantly influence the total result.

Table 7.9 Updated prices of major crops.

	Updated Prices per tonne	Source
Sunflower	273	FAOSTAT € (2001)
Wheat	137	IFS € (2003)
Potato	113	FAOSTAT € (2001)
Rice	200	IFS € (2003)
Rye	99	FAOSTAT € (2001)
Oats	132	FAOSTAT € (2001)
Tobacco	2895	IFS € (2003)
Barley	93	IFS € (2003)
Sugar beet	64	FAO € (2002)

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8 Other Impacts: Global Warming

8.1 Introduction

Climate change is one of the most prominent environmental problems of today. Its impacts are far-reaching in space and in time, while prosperity and fossil fuel use are close entwined. For the estimation of external costs for climate change, two methodologies are followed. On the one hand, models are applied to estimate damage costs occurring due to impacts from climate change and, on the other hand, avoidance costs are estimated as an equivalent for the preferences followed when focussing on a target, e.g. based on the motivation to follow the path to sustainable development.

8.2 The Model

The Climate Framework for Uncertainty, Negotiation and Distribution (FUND) has been developed over a period of several years. Thus, in different projects, different versions of FUND were used. For the currently running EU Integrated Project NEEDS, estimates will be based on FUND version 2.8. Parts of the model go back to version 1.6 (see Tol, 1997, 1999a-e, 2001, 2002a). Other parts go back to version 2.0 and version 2.4 (Tol and Heinzow, 2003). The main changes are that the current version of the model has 16 rather than 9 regions, that the impacts of climate change on diarrhoea are included, and that methane and nitrous oxide are included as options for greenhouse gas emission abatement.

Essentially, *FUND* consists of a set of exogenous scenarios and endogenous perturbations, specified for nine major world-regions, namely the United States of America, Canada, Western Europe, Japan and South Korea, Australia and New Zealand, Central and Eastern Europe, the former Soviet Union, the Middle East, Central America, South America, South Asia, Southeast Asia, China, North Africa, Sub-Saharan Africa, and Small Island States. See Table 8.1.

The model runs from 1950 to 2300, in time steps of a year. The prime reason for extending the simulation period into the past is the necessity to initialise the climate change impact module. In *FUND*, some climate change impacts are assumed to depend on the impact of the year before, so as to reflect the process of adaptation to climate change. Without a proper initialisation, climate change impacts are thus misrepresented in the first decades. Scenarios for the period 1950-1990 are based on historical observation, viz. the *IMAGE* 100-year database (Batjes and Goldewijk, 1994). The period 1990-2100 is based on the *FUND* scenario, which lies somewhere in between the IS92a and IS92f scenarios (Leggett *et al.*, 1992). Note that the original IPCC scenarios had to be adjusted to fit *FUND*'s sixteen regions and yearly time-step.

Table 8.1 The regions in FUND.

Acronym	Name	Countries
USA	USA	United States of America
CAN	Canada	Canada
WEU	Western Europe	Andorra, Austria, Belgium, Cyprus, Denmark, Finland, France, Germany, Greece, Iceland, Ireland, Italy, Liechtenstein, Luxembourg, Malta, Monaco, Netherlands, Norway, Portugal, San Marino, Spain, Sweden, Switzerland, United Kingdom
JPK	Japan & South Korea	Japan, South Korea
ANZ	Australia & New Zealand	Australia, New Zealand
CEE	Central and Eastern Europe	Albania, Bosnia and Herzegovina, Bulgaria, Croatia, Czech Republic, Hungary, FYR Macedonia, Poland, Romania, Slovakia, Slovenia, Yugoslavia
FSU	Former Soviet Union	Armenia, Azerbaijan, Belarus, Estonia, Georgia, Kazakhstan, Latvia, Lithuania, Moldova, Russia, Tajikistan, Turkmenistan, Ukraine, Uzbekistan
MDE	Middle East	Bahrain, Iran, Iraq, Israel, Jordan, Kuwait, Lebanon, Oman, Qatar, Saudi Arabia, Syria, Turkey, United Arab Emirates, West Bank and Gaza, Yemen
CAM	Central America	Belize, Costa Rica, El Salvador, Guatemala, Honduras, Mexico, Nicaragua, Panama
SAM	South America	Argentina, Bolivia, Brazil, Chile, French Guiana, Guyana, Paraguay, Peru, Suriname, Uruguay, Venezuela
SAS	South Asia	Afghanistan, Bangladesh, Bhutan, India, Nepal, Pakistan, Sri Lanka
SEA	Southeast Asia	Brunei, Cambodia, East Timor, Indonesia, Laos, Malaysia, Myanmar, Papua New Guinea, Philippines, Singapore, Taiwan, Thailand, Vietnam
CHI	China plus	China, Hong Kong, North Korea, Macau, Mongolia
NAF	North Africa	Algeria, Egypt, Libya, Morocco, Tunisia, Western Sahara
SSA	Sub-Saharan Africa	Angola, Benin, Botswana, Burkina Faso, Burundi, Cameroon, Cape Verde, Central African Republic, Chad, Congo-Brazzaville, Congo-Kinshasa, Cote d'Ivoire, Djibouti, Equatorial Guinea, Eritrea, Ethiopia, Gabon, Gambia, Ghana, Guinea, Guinea- Bissau, Kenya, Lesotho, Liberia, Madagascar, Malawi, Mauritania, Mozambique, Namibia, Niger, Nigeria, Rwanda, Senegal, Sierra Leone, Somalia, South Africa, Sudan, Swaziland, Tanzania, Togo, Uganda, Zambia, Zimbabwe
SIS	Small Island States	Antigua and Barbuda, Aruba, Bahamas, Barbados, Bermuda, Comoros, Cuba, Dominica, Dominican Republic, Fiji, French Polynesia, Grenada, Guadeloupe, Haiti, Jamaica, Kiribati, Maldives, Marshall Islands, Martinique, Mauritius, Micronesia, Nauru, Netherlands Antilles, New Caledonia, Palau, Puerto Rico, Reunion, Samoa, Sao Tome and Principe, Seychelles, Solomon Islands, St Kitts and Nevis, St Lucia, St Vincent and Grenadines, Tonga, Trinidad and Tobago, Tuvalu, Vanuatu, Virgin Islands

The period 2100-2300 is based on extrapolation of the population, economic and technological trends in 2050-2100, that is, a gradual shift to a steady state of population, economy and technology.

The scenarios concern the rate of population growth, economic growth, autonomous energy efficiency improvements, the rate of decarbonisation of energy use (autonomous carbon efficiency improvements), and emissions of carbon dioxide from land use change, methane and nitrous oxide.

The scenarios of economic and population growth are perturbed by the impact of climatic change. Population decreases with increasing climate change-related deaths that result from changes in heat stress, cold stress, malaria, and tropical cyclones. Heat and cold stress are assumed to have an effect only on the elderly, non-reproductive population. In contrast, the other sources of mortality also affect the number of births. Heat stress only affects the urban population. The share of the urban population among the total population is based on the World Resources Databases (WRI, 2000). It is extrapolated based on the statistical relationship between urbanisation and per-capita income which are estimated from a cross-section of countries in 1995. Climate-induced migration between the regions of the world also causes the population sizes to change. Immigrants are assumed to assimilate immediately and completely with the respective host population. The tangible impacts are dead-weight losses to the economy. Consumption and investment are reduced without changing the savings rate. Thus, climate change reduces the long-term economic growth, although for the short-term consumption is particularly affected. Economic growth is also reduced by carbon dioxide abatement measures. The energy intensity of the economy and the carbon intensity of the energy supply autonomously decrease over time.

The endogenous parts of *FUND* consist of carbon dioxide emissions, the atmospheric concentrations of carbon dioxide, methane and nitrous oxide, the global mean temperature, and the impact of climate change on coastal zones, agriculture and forestry, energy consumption, water resources, natural ecosystems and human health. The impact module is described in more detail in the next section. *FUND* uses simple models to represent all these components; each simple model is calibrated either to more complex models or to data; *FUND* as a whole has no match, either model or observations.

Carbon dioxide emissions are calculated on the basis of the Kaya identity:

$$M_{r,t} = \frac{M_{r,t}}{E_{r,t}} \frac{E_{r,t}}{Y_{r,t}} \frac{Y_{r,t}}{P_{r,t}} P_{r,t} := \psi_{r,t} \varphi_{r,t} Y_{r,t} \tag{8.1}$$

where M denotes emissions, E energy use, Y gross domestic product, and P population; r indexes region, t time. The carbon intensity of energy use ψ and the energy intensity of production φ follow from:

$$\psi_{r,t} = g^{\psi}_{r,t-1}\psi_{r,t-1} - \alpha\tau^{\psi}_{r,t-1} \tag{8.2}$$

and

$$\varphi_{r,t} = g^{\varphi}_{r,t-1}\varphi_{r,t-1} - \alpha\tau^{\varphi}_{r,t-1} \tag{8.3}$$

where τ is policy intervention, g is autonomous technological change, and α is a parameter. Policy affects emissions via

$$M_{r,t} = \left(\psi_{r,t} - \chi^{\psi}_{r,t}\right)\left(\varphi_{r,t} - \chi^{\varphi}_{r,t}\right)Y_{r,t} \tag{8.4}$$

$$\chi^{\psi}_{r,t} = \kappa_{\psi}\chi_{r,t-1} + (1-\alpha)\tau^{\psi}_{r,t-1} \tag{8.5}$$

and

$$\chi^{\varphi}_{r,t} = \kappa_{\varphi}\chi_{r,t-1} + (1-\alpha)\tau^{\varphi}_{r,t-1} \tag{8.6}$$

Thus, the parameter $0 < \alpha < 1$ governs which part of emission reduction is *permanent* (reducing carbon and energy intensities) and which part of emission reduction is *temporary* (reducing energy consumptions and carbon emissions), fading at a rate of $0 < \kappa < 1$. Alternatively, one can interpret the difference between permanent and temporary emission reduction as affecting commercial technologies and capital stocks, respectively. The behaviour of the emission reduction module is similar to the models of Grubb *et al.* (1995), Ha-Duong *et al.* (1997) and Hasselmann *et al.* (1997).

The costs of emission reduction are given by

$$\frac{C_{r,t}}{Y_{r,t}} = \frac{\beta_{r,t}\tau^2_{r,t}}{H_{r,t}H^g_t} \tag{8.7}$$

The parameter β follows from

$$\beta_{r,t} = 2.24 - 0.24\sqrt{\frac{M_{r,t}}{Y_{r,t}} - \min_s \frac{M_{s,t}}{Y_{s,t}}} \tag{8.8}$$

That is, emission reduction is relatively expensive for the region that has the lowest emission intensity. The calibration is such that a 10% emission reduction would cost 2.24% of GDP. Emission reduction is relatively cheap for regions with high emission intensities. The thought is that emission reduction is cheap in countries that use a lot of energy and rely heavily on fossil fuels, while other countries use less energy and less fossil fuels. The model was calibrated to the results reported in Hourcade *et al.* (1996).

The regional and global knowledge stocks follow from

$$H_{r,t} = H_{r,t-1}\sqrt{1+\gamma_R\tau_{r,t-1}} \tag{8.9}$$

and

$$H^G_t = H^G_{t-1}\sqrt{1+\gamma_G\tau_{r,t}} \tag{8.10}$$

Knowledge accumulates with emission abatement. The parameter γ determines which part of the knowledge is kept within the region, and which part spills over to other regions as well. In the base case, γ_R=0.9 and γ_G=0.1. The model is similar in structure and numbers to that of Goulder and Schneider (1999) and Goulder and Mathai (2000).

The costs of methane and nitrous oxide emission reduction are based on the analysis of the US EPA. They report supply curves of emission reduction, stating how much can be abated at a certain price. First, these supply curves were shifted to exclude negative costs. Note that this increases costs. Second, emission reductions were expressed as fractions of baseline emissions. Third, total emission reduction costs (the area under the supply curve) were calculated, and expressed as a fraction of GDP. Fourth, the regional results of the EPA analysis were attributed to the *FUND* regions. Fifth, the bottom-up curve was approximated with a smooth exponential function. Sixth, the exponential curve was approximated with a quadratic curve. Note that this decreases costs. Table 8.2 shows the parameters for methane, Table 8.3 for nitrous oxide. The quadratic cost curve has the advantage that both costs and marginal costs are zero at zero emission reduction. The exponential cost curve has total costs equal to zero at zero emission reduction, but marginal costs are greater than zero. This implies that, for a low carbon price, methane and nitrous oxide emission reduction are zero, while carbon dioxide emission reduction is not. On the other hand, large emission reduction is cheaper with the quadratic specification than with the exponential one. Nonetheless, we prefer the quadratic specification to the exponential one.

Table 8.2 Parameters of the methane emission reduction cost curve; the 67% confidence interval is given in brackets.

	Quadratic			Exponential - constant			Exponential - exponent		
USA	5.74E-04	(4.15E-04	7.90E-04)	5.43E-06	(4.44E-06	6.64E-06)	10.28	(9.66	10.90)
CAN	1.20E-03	(8.70E-04	1.64E-03)	7.69E-06	(6.30E-06	9.37E-06)	12.49	(11.75	13.23)
WEU	3.71E-04	(2.34E-04	5.80E-04)	1.82E-06	(1.37E-06	2.43E-06)	14.27	(13.10	15.45)
JPK	1.27E-04	(8.75E-05	1.84E-04)	4.19E-07	(3.32E-07	5.29E-07)	17.43	(16.23	18.63)
ANZ	4.12E-03	(3.03E-03	5.57E-03)	1.25E-05	(1.03E-05	1.51E-05)	18.18	(17.14	19.21)
EEU	3.90E-03	(2.81E-03	5.38E-03)	3.13E-05	(2.56E-05	3.83E-05)	11.17	(10.49	11.85)
FSU	8.87E-03	(7.49E-03	1.05E-02)	8.51E-05	(7.65E-05	9.46E-05)	10.21	(9.89	10.52)
MDE	6.32E-03	(4.86E-03	8.19E-03)	1.26E-05	(1.07E-05	1.49E-05)	22.38	(21.29	23.47)
CAM	3.65E-03	(2.87E-03	4.62E-03)	1.30E-05	(1.12E-05	1.51E-05)	16.77	(16.03	17.52)
SAM	2.75E-02	(1.81E-02	4.14E-02)	4.07E-06	(3.14E-06	5.27E-06)	82.24	(75.89	88.58)
SAS	3.16E-02	(2.43E-02	4.08E-02)	2.51E-05	(2.13E-05	2.95E-05)	35.45	(33.74	37.16)
SEA	1.43E-02	(1.06E-02	1.91E-02)	1.94E-05	(1.62E-05	2.33E-05)	27.15	(25.66	28.65)
CHI	1.26E-02	(9.50E-03	1.67E-02)	3.18E-05	(2.67E-05	3.80E-05)	19.93	(18.88	20.97)
MAF	1.43E-02	(1.06E-02	1.91E-02)	1.94E-05	(1.62E-05	2.33E-05)	27.15	(25.66	28.65)
SSA	1.43E-02	(1.06E-02	1.91E-02)	1.94E-05	(1.62E-05	2.33E-05)	27.15	(25.66	28.65)
SIS	1.43E-02	(1.06E-02	1.91E-02)	1.94E-05	(1.62E-05	2.33E-05)	27.15	(25.66	28.65)

Table 8.3 Parameters of the nitrous oxide emission reduction cost curve; the 67% confidence interval is given in brackets.

	Quadratic			Exponential - constant			Exponential - exponent		
USA	2.14E-05	(1.91E-05	2.39E-05)	1.36E-08	(1.29E-08	1.45E-08)	39.61	(38.56	40.65)
CAN	6.92E-05	(6.29E-05	7.60E-05)	1.62E-08	(1.54E-08	1.70E-08)	65.33	(63.88	66.78)
WEU	7.26E-06	(6.60E-06	7.98E-06)	1.97E-08	(1.88E-08	2.08E-08)	19.18	(18.75	19.60)
JPK	5.32E-07	(3.21E-07	8.57E-07)	9.54E-09	(7.38E-09	1.23E-08)	7.46	(6.60	8.33)
ANZ	2.08E-04	(1.89E-04	2.29E-04)	4.62E-09	(4.39E-09	4.86E-09)	212.40	(207.68	217.11)
EEU	9.39E-05	(8.89E-05	9.93E-05)	8.35E-08	(7.91E-08	8.83E-08)	33.53	(33.53	33.53)
FSU	1.05E-05	(1.00E-05	1.10E-05)	1.94E-08	(1.91E-08	1.98E-08)	23.25	(22.91	23.60)
MDE	1.05E-05	(1.00E-05	1.10E-05)	1.94E-08	(1.91E-08	1.98E-08)	23.25	(22.91	23.60)
CAM	2.35E-04	(2.19E-04	2.53E-04)	2.00E-08	(1.89E-08	2.13E-08)	108.39	(107.83	108.95)
SAM	1.05E-05	(1.00E-05	1.10E-05)	1.94E-08	(1.91E-08	1.98E-08)	23.25	(22.91	23.60)
SAS	5.64E-04	(5.29E-04	6.01E-04)	1.71E-07	(1.62E-07	1.80E-07)	57.44	(57.14	57.74)
SEA	2.55E-15	(2.16E-15	3.01E-15)	4.72E-18	(4.12E-18	5.40E-18)	23.25	(22.91	23.60)
CHI	2.16E-05	(2.02E-05	2.30E-05)	1.42E-07	(1.35E-07	1.50E-07)	12.32	(12.26	12.39)
MAF	1.05E-05	(1.00E-05	1.10E-05)	1.94E-08	(1.91E-08	1.98E-08)	23.25	(22.91	23.60)
SSA	1.05E-05	(1.00E-05	1.10E-05)	1.94E-08	(1.91E-08	1.98E-08)	23.25	(22.91	23.60)
SIS	1.05E-05	(1.00E-05	1.10E-05)	1.94E-08	(1.91E-08	1.98E-08)	23.25	(22.91	23.60)

Methane and nitrous oxide are taken up in the atmosphere and then geometrically depleted:

$$C_t = C_{t-1} + \alpha E_t - \beta(C_{t-1} - C_{pre}) \tag{8.11}$$

where C denotes concentration, E emissions, t year, and *pre* pre-industrial. Table 8.4 displays the parameters for both gases. Equation (8.11) is a simplified representation of the relevant atmospheric chemistry. Particularly, the atmospheric lifetime is not constant but depends on the concentrations and emissions of other chemical species.

Table 8.4 Parameters of Eq. (8.11). Source: Shine *et al.* (1990).

Gas	α[a]	β[b]	Pre-industrial concentration
Methane (CH_4)	0.3597	1/8.6	790 ppb
Nitrous oxide (N_2O)	0.2079	1/120	285 ppb

[a] The parameter α translates emissions (in million metric tonnes of CH_4 or N_2O) into concentrations (in parts per billion by volume).
[b] The parameter β determines how fast concentrations return to their pre-industrial (and assumedly equilibrium) concentrations; 1/β is the atmospheric lifetime (in years) of the gases.

The carbon cycle is a five-box model[19]:

[19] The boxes have no physical representation. Rather, the model is a Green's function approximation to a

$$Box_{i,t} = \rho_i \, Box_{i,t-1} + 0.000471 \alpha_i \, E_t \qquad (8.12)$$

with

$$C_t = \sum_{i=1}^{5} \alpha_i \, Box_{i,t} \qquad (8.13)$$

where α_i denotes the fraction of emissions E (in million metric tonnes of carbon) that is allocated to box i (0.13, 0.20, 0.32, 0.25 and 0.10, respectively) and ρ the decay rate of the boxes (ρ = exp(-1/lifetime), with lifetimes infinity, 363, 74, 17 and 2 years, respectively). Thus, 13% of total emissions remains forever in the atmosphere, while 10% is – on average – removed in two years. The model is due to Maier-Reimer and Hasselmann (1987), its parameters to Hammitt *et al.* (1992). It assumes, incorrectly, that the carbon cycle is independent of climate change. Carbon dioxide concentrations are measured in parts per million by volume.

Radiative forcing for carbon dioxide, methane and nitrous oxide are based on Shine *et al.* (1990). The global mean temperature T is governed by a geometric build-up to its equilibrium (determined by radiative forcing RF), with a lifetime of 50 years. In the base case, global mean temperature rises in equilibrium by 2.5°C for a doubling of carbon dioxide equivalents, so:

$$T_t = \left(1 - \frac{1}{50}\right) T_{t-1} + \frac{1}{50} \frac{2.5}{6.3 \ln(2)} RF_t \qquad (8.14)$$

Global mean sea level is also geometric, with its equilibrium determined by the temperature and a lifetime of 50 years. These lifetimes result from a calibration to the best guess temperature and sea level for the IS92a scenario of Kattenberg *et al*. (1996).

The basis of the climate impact module is fully described in Tol (2002a,b). The impact module has two units of measurement: people and money. People can die prematurely and migrate. These effects, like all other impacts, are monetised. Damage can be due to either the rate of change or the level of change. Benchmark estimates can be found in Table 8.5.

Impacts of climate change on energy consumption, agriculture, and cardiovascular and respiratory diseases explicitly recognise that there is a climate optimum. The climate optimum is determined by a mix of factors, including physiology and behaviour. Impacts are positive or negative depending on whether climate is moving towards or

complex ocean carbon-cycle model, with five characteristic lifetimes.

away from that optimum climate. Impacts are larger if the initial climate is further away from the optimum climate. The optimum climate concerns the potential impacts. Actual impacts lag behind potential impacts, depending on the speed of adaptation. The impacts of not being fully adapted to the new climate are always negative. See Tol (2002b).

Table 8.5 Estimated impacts of a 1°C increase in the global mean temperature. Standard deviations are given in brackets (source: Tol, 2002b).

	Billion dollar		percent of GDP	
OECD-A	175	(107)	3.4	(2.1)
OECD-E	203	(118)	3.7	(2.2)
OECD-P	32	(35)	1.0	(1.1)
CEE&FSU	57	(108)	2.0	(3.8)
ME	4	(8)	1.1	(2.2)
LA	-1	(5)	-0.1	(0.6)
S&SEA	-14	(9)	-1.7	(1.1)
CPA	9	(22)	2.1	(5.0)
AFR	-17	(9)	-4.1	(2.2)

Other impacts of climate change, on coastal zones, forestry, unmanaged ecosystems, water resources, malaria, dengue fever and schistosomiasis, are modelled as simple power functions. Impacts are either negative or positive, but do not change sign. See Tol (2002b). Diarrhoea follows a similar logic. The number of additional diarrhoea deaths D^d is given by

$$D_{r,t}^{d} = \mu_r^d P_{r,t} \left(\frac{y_{r,t}}{y_{r,0}} \right)^{\varepsilon} \left(T_{r,t}^{\eta} - T_{r,0}^{\eta} \right) \tag{8.15}$$

where P denotes population, y per capita income, and T regional temperature; μ is the baseline mortality, ε (see below) and η=1.14 (with a standard deviation of 0.51) are parameters; r indexes region, and t time. Equation (8.15) was estimated based on the WHO Global Burden of Diseases data (http://www.who.int/health_topics/global_burden_of_disease/en/). Diarrhoea morbidity has the same equation as mortality, but with η=0.70 (0.26). Table 8.6 shows benchmark estimates.

Vulnerability to climate change alters with population growth, economic growth and technological progress. Some systems are expected to become more vulnerable, such as water resources (with population growth), heat-related disorders (with urbanisation), and ecosystems and health (with higher per capita incomes). Other systems are projected to become less vulnerable, such as energy consumption (with technological progress), agriculture (with economic growth) and vector- and water-borne diseases (with improved health care) (cf. Tol, 2002b). Vector-borne diseases fall with economic growth, using a per capita income elasticity of -2.65 with a standard deviation of 0.69.[20] The income elasticity of diarrhoea mortality is -1.58 (0.23), for diarrhoea morbidity -0.42 (0.12). These elasticities were estimated based on the WHO Global Burden of Diseases data (http://www.who.int/health_topics/global_burden_of_disease/en/).

Table 8.6 Diarrhoea mortality and morbidity due to a 2.5˚C global warming.

Region	Population[a]	Mortality[b]	Morbidity[c]	ΔT[d]	Additional Mortality[e]		Additional Morbidity[f]	
USA	278357	0.041	1.704	3.0	40	(23 70)	1019	(767 1354)
CAN	31147	0.041	1.704	3.7	6	(3 11)	132	(94 185)
WEU	388581	0.015	0.632	2.8	18	(11 31)	506	(387 662)
JPK	173558	0.009	0.166	2.6	5	(3 8)	57	(44 73)
ANZ	22748	0.001	0.083	2.4	0	(0 0)	3	(3 4)
EEU	121191	0.018	0.847	2.9	7	(4 13)	217	(164 287)
FSU	291538	0.122	6.735	3.2	135	(74 244)	4443	(3279 6020)
MDE	237590	0.030	0.166	2.9	24	(14 41)	83	(63 109)
CAM	135222	0.162	0.643	2.2	54	(36 81)	151	(123 185)
LAM	345779	0.168	0.650	2.1	138	(94 202)	381	(313 463)
SAS	1366902	0.229	0.896	2.3	798	(526 1212)	2171	(1755 2687)
SEA	522462	0.135	0.631	1.8	136	(102 182)	492	(424 571)
CHI	1311659	0.033	0.401	3.0	150	(86 261)	1122	(846 1488)
MAF	143482	0.415	0.990	2.9	197	(116 337)	296	(225 389)
SSA	637887	3.167	5.707	2.2	4958	(3321 7404)	6306	(5141 7737)
SIS	44002	0.252	1.092	1.9	23	(17 31)	75	(63 88)

[a] Thousands of people, 2000.
[b] Deaths per thousand people.
[c] Years of life diseased per thousand people.
[d] Regional temperature change for a 2.5˚C global warming.
[e] Additional deaths, thousands of people (67% confidence interval in brackets).
[f] Additional years of life diseased, thousands (67% confidence interval in brackets).

8.3 Marginal Cost Estimates

Marginal costs of carbon dioxide are estimated as follows. First, a base run is made with the model. Second, a perturbed run is made in which one million metric tonnes of carbon are added to the atmosphere for the period 2000-2009. In both runs, relative impacts, GDP and population are saved. Marginal costs are estimated using:

[20] In previous model versions, vector-borne diseases fall linearly to zero at an annual per capita income of $3100, based on Tol and Dowlatabadi (2001). Increased data availability allowed us to move away from this simple representation.

$$\frac{\sum_{r=1}^{9}\sum_{t=0}^{150}\left(\frac{D_{r,t}^{P}}{Y_{r,t}^{P}}-\frac{D_{r,t}^{B}}{Y_{r,t}^{B}}\right)\frac{Y_{r,t}^{B}}{(1+\rho+g_{r,t}^{B})^{t}}(\$)}{10000000(tC)} \tag{8.16}$$

where D is monetised damage; Y is GDP, g is the growth rate of per capita income; ρ is the pure rate of time preference; the subscript t is time; and the superscript denotes base (B) or perturbed (P) run. That is, the change in *relative* impacts is evaluated against the baseline economic growth – this is to avoid the complications of differential effects on the economic growth path (see Fankhauser and Tol, 2001, for a discussion). Impacts are discounted using the standard neo-classical discount rate, viz., the sum of the pure rate of time preference and the growth rate of per capita consumption.

Table 8.7 shows some sample calculations for the marginal damage costs of carbon dioxide. The numbers are in the same range as for previous version of the FUND model, and well in line with the literature (Tol, 2005).

Table 8.7 Marginal damage costs of climate change ($/t C) with and without a thermohaline circulation collapse (THC), for three alternative discount rates (0, 1 and 3 per cent pure rate of time preference), for simple summation (SS) and equity weighing (EW).

Discount rate	0 %		1 %		3 %	
	SS	EW	SS	EW	SS	EW
No THC	79.0	170.0	25.2	94.1	5.1	45.1
THC	75.6	167.8	24.4	93.6	5.0	45.0

Tol (2005) contains the most comprehensive review of the marginal damage costs of carbon dioxide to date. Table 8.8 reproduces his key findings. Given the large uncertainty, the median is the best measure of central tendency. Depending on the pure rate of time preference (PRTP) one wants to use, the marginal damage cost is either $7/tC or $33/tC.[21]

Table 8.8 The marginal costs of carbon dioxide emissions ($/tC).

	Mode	Mean	5%	10%	Median	90%	95%
All	1.5	93	-10	-2	14	165	350
PRTP=3% only	1.5	16	-6	-2	7	35	62
PRTP=1% only	4.7	51	-14	-2	33	125	165
PRTP≤ 0% only	6.9	261	-24	-2	39	755	1610

[21] In 1995 prices.

8.4 Evaluation of Policy Decisions

The damage cost estimates presented in the previous sections give a rather broad and uncertain range of results. Furthermore it remains unclear to what extent these data give a complete picture of the total impact, as a wide number of impacts are not included and for those that are included, uncertainties are large, both for quantification of effects and for the valuation.

Given the uncertainties and incompleteness inherent in these estimates, one can argue that the balancing of costs and benefits in negotiations over targets and/or policy measures may offer a complementary view on how society values the benefits of the first steps in CO_2 control. Therefore, in ExternE (see ExternE, 2004) two approaches based on revealed preferences have been explored. The first is to estimate revealed preferences based on policy targets. A second approach is based on public preferences as revealed in referenda related to energy questions in Switzerland.

8.4.1 Selection of policy targets and their interpretation

The first issue is to select the most relevant policy targets and to interpret the arguments used in the negotiations leading up to these decisions. The main target at the EU level is the Kyoto Protocol of 1997, which has been ratified by the EU and its member states in 2002. The European Climate Change Programme of 2000 elaborates a roadmap to translate this target into proposals. The Kyoto Protocol defines the target for the EU to reduce greenhouse gas emissions by 8 % by 2008-2012 compared to 1990 emissions, for the EU 15 as a whole. The Protocol itself, however, does not indicate how the target should be achieved. This is an important question because the costs of meeting Kyoto will depend on the policy mechanism chosen.

The policies in Europe related to climate change show a tradition of looking for a balance between (i) dividing the target between member states and sectors, leaving it open to member states and/or sectors to look for measures to achieve these targets; and (ii) deciding at EU level on concrete policy measures, sector-specific or cross-sector (e.g. a CO_2 tax or EU-wide emission trading system).

The final decisions still show a mixture of these approaches. First, the EU has developed differentiated targets for each member country in order to share the economic burden of climate protection equitably. This so-called "burden-sharing" agreement between EU governments lays down differentiated emission limits for each member state with the aim of ensuring that the EU meets its overall 8% reduction commitment under the Protocol. The limits are expressed in terms of percentages by which Member states must reduce, or in some cases may hold or increase, their emissions compared with the base year level (1990). The differentiated targets for countries reflect the fact that costs and the capacities to carry these costs may differ, as well as society's willingness to take early action. The EU member states have to develop National Allocation Plans (NAPs) to indicate how they will achieve these emissions reductions. Second, a combination of measures at European and national level is required, including flexible mechanisms like the EU Emissions Trading Scheme (EU ETS) which started on the first of January 2005. The objective of the latter is to allow for a cost efficient reduction of CO_2-emissions for big industrial energy users. In addition, additional measures and targets will be required, e.g. for transportation and household sectors, both at European and national level. Third, countries have the possibilities to meet their emission reduction targets by using the so-called flexible mechanisms or Kyoto Mechanisms, like Joint Implementation or International Emissions Trading.

As part of the preparation of the Kyoto Protocol, the potential for CO_2 emission reductions in the EU and their costs were well documented. Therefore, it is fair to say that in preparing and implementing the Kyoto agreement, these costs were balanced against the benefits. There are several limitations for the use of this information as a revealed preference from policy decisions.

First, the real preferences will be revealed in the policies implemented, rather than in the phase of setting targets. This would however require a careful assessment of all national plans to see which policy measures will be implemented, to see the real 'willingness to pay to combat global warming' from policy and decision makers. This work can only be done when the final plans are available and accepted by the EC. Therefore, this analysis is based on more generic information on reduction costs per ton of CO_2. For the interpretation of the data, we will use some additional information on policy plans, etc.

Second, the main benefit of the first steps towards CO_2 control is not only a reduction in damages from global warming, but they also contribute to build a worldwide strategy to combat global warming. In this context, the benefits of meeting the Kyoto target (expressed per ton of CO_2) may have a multiplicator effect, which is not reflected in the figures used for the decision-making.

Third, controlling CO_2 emissions will result in benefits in other areas including air quality and energy security. These so-called no-regret benefits have not been documented in detail and are not accounted for.

The discussion and data mainly focused on one greenhouse gas, i.e. CO_2, whereas the Protocol covers all greenhouse gases.

8.4.2 A shadow price for CO_2 emissions in Europe

In the policy process leading to the adoption of the European Climate Change Programme and the proposal for a directive on CO_2 trading mechanisms, several studies on the costs of meeting these targets were executed, mostly using energy-economic models. The latest studies for the EU suggest that, under a full flexibility EU-wide allocation of least cost sectoral objectives, the marginal abatement cost will be €20 per tonne. These estimates are based both on top-down and bottom up approaches. A recent review showed that this estimate is in the middle of the wider range of estimates, both from studies and from starting or experimental CO_2-trading schemes (Downing and Watkiss, 2003). However, when each member state tries to fulfil its objectives on its own, the marginal cost for Belgium will increase up to €90 per tonne CO_2 (Blok *et al.*, 2001). On the other hand, allowing some kind of trading outside the EU may lower the compliance costs to perhaps €5 per tonne. Consequently, most studies take a figure close to this €20 per tonne of CO_2 as the marginal abatement costs, and a proxy for society's willingness to pay, for Europe. This number is also well below the penalty set in the emission trading scheme (€40 per tonne of CO_2 for the first 3 years), and can be seen as an upper limit for this shadow price.

From a theoretical point of view, there are reasons to argue for higher or lower numbers, but our analysis shows that they are no better estimates than the range of €5-20/t CO_2.

As a number of countries accepted stricter emission reduction targets and took earlier unilateral actions to limit CO_2 emissions, and as studies indicated that they would also require the more costly emission reductions, one can argue that the WTP in some countries may be higher. Given the differences in emission reduction targets and the costs to meet these targets, there are good reasons to argue for country-specific shadow prices for CO_2. Consequently, some propose a national shadow price for CO_2. As an example, from analysis of policy targets for the Netherlands and national costs estimates, a shadow price of €50 per tonne of CO_2 equivalent is proposed (Davidson *et al.,* 2002).

Although the marginal abatement costs for reaching the objectives are available per country, these cannot be taken as a proxy for society's WTP per country, unless more evidence to support such values is available. One may also argue that, for many member states, their recent record in emission trends does not support the idea of a high WTP, as most member states lag behind a theoretical linear Kyoto target path (EEA, 2004). Second, a recent overview of draft national plans illustrated that a number of countries will need the cheaper Kyoto flexible mechanisms to reach the Kyoto target (Ecofys, 2004). The costs of using flexible mechanisms will be lower, but it is still unclear to what extent these mechanisms will be used and what the marginal prices are likely to be.

One can argue that the market prices for CO_2 emission allowances under the EU ETS inform us about the real 'shadow price' for CO_2 and the real WTP from policy makers. It is hard to estimate to what extent a shadow price will be reflected in real life decision making in the sectors because it is very unclear to all potential actors in the market how this market will develop. Indeed, as the industries subject to the EU ETS will receive emission allowances (grand-fathering) based on the national allocation plans, national governments will make some cost-benefit considerations in controlling CO_2 emissions in sectors subject to the ETS or in other sectors. It is unlikely that a future market price for CO_2 emission allowances also has been taken into consideration, because the development of this market and future prices are very unclear. In the long run however, if the EU ETS scheme develops into a real market, this could be a better indicator than the current data from technical-economic studies.

It may be argued that the real WTP will be lower than the range suggested above, because policy makers are aware of benefits in other areas like energy saving or air pollution. Although this argument is true, there are no data to correct for this potential effect. This remark is in support of choosing a best estimate on the lower side of the range.

8.4.3 Application of shadow prices for CO_2 and greenhouse gases

An assessment of the costs for achieving Kyoto targets can be interpreted as a proxy for society's willingness-to-pay for early action against global warming. For assessing technologies and fuel cycles in the mid to long-term, the best estimate is between €5-20/t of CO_2, with the higher range reflecting the costs if emissions are controlled within Europe. By extension, it can be applied to all greenhouse gases.

This shadow price for CO_2, based on the marginal abatement costs to meet the Kyoto target, reflects the CO_2 efficiency of energy technologies or fuel cycles. Those that are more efficient will be given credit for this benefit, which allows European society and economies to save costs for meeting the Kyoto target.

When applying this range, some remarks have to be considered. First, it needs to be evaluated on a case-by-case base whether this figure is applicable and whether some kind of CO_2-externality has already been internalised. Within the sectors subject to the emission-trading regime (e.g. electricity generation), a price incentive that reflects CO_2-efficiency will be installed from 2005 onwards. *A priori*, however, one cannot decide to what extent the EU ETS scheme will develop into an active market. The average electricity price for consumers however, will not contain a price signal that reflects overall CO_2 efficiency. When comparing technologies on a full fuel cycle basis, emissions outside the EU are unlikely to be subject to price incentives that reflect CO_2-efficiency.

Second, depending on the context, sector or country-specific marginal abatement costs may be better than the European marginal abatement cost. This is the case if the shadow price needs to reflect the contribution of that technology or fuel cycle to a specific target at national or sectoral level. This will be especially the case for decisions with a short time impact, and limited to a specific sector or country. The same reasoning goes for shadow prices for other greenhouse gases. On the other hand, if the objective is to reflect some overall shadow price for making (small) progress towards controlling greenhouse gases, the overall marginal European marginal abatement cost for CO_2 is a better proxy and can be applied to all greenhouse gases. This will especially be the case for decisions with a longer time horizon, and a cross-sector or cross-border impact.

8.4.4 More ambitious emission reduction

Figure 8.1 shows the net present value of the loss of consumption due to emission reduction in the OECD, Eastern Europe and the former Soviet Union (EEFSU), and developing countries (DC). With all three gases, meeting the target of 4.5 Wm^{-2} would cost \$32.9 trillion. The tax on carbon dioxide emissions could be as high as \$350/tC.

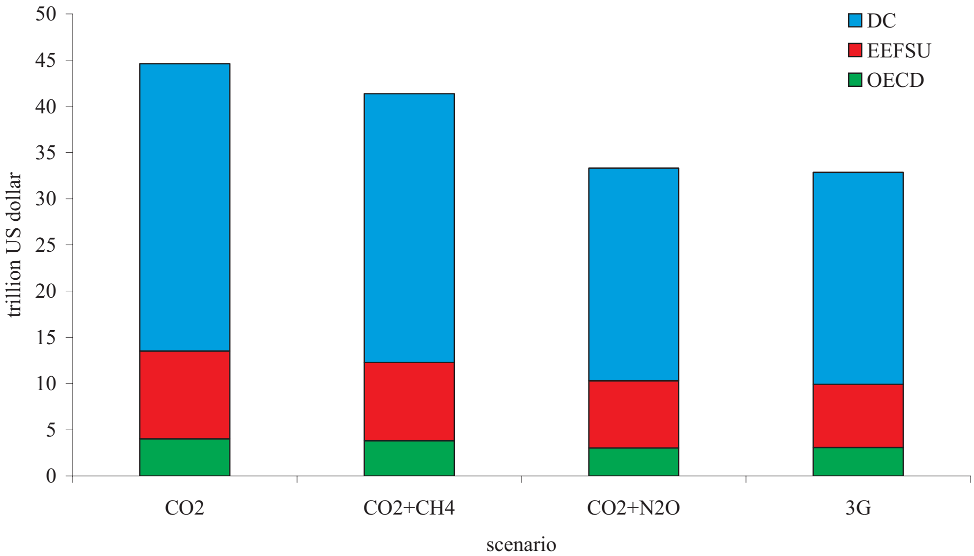

Figure 8.1 The net present value of consumption losses due to alternative policies to keep anthropogenic radiative forcing below 4.5 Wm^{-2}, viz. only CO_2 emission reduction, CO_2 and CH_4 emission reduction, CO_2 and N_2O emission reduction, and emission reduction with all three gases (3G).

These numbers are so high because the target is so strict. This target roughly corresponds to the 2°C target of the European Union. With CO_2 emission reduction only, the costs would rise to \$44.6 trillion. Methane and nitrous oxide emission reduction thus reduce costs by some 26%, substantially less than reported elsewhere. Most of the cost saving is due to nitrous oxide. Without N_2O, the costs would be \$41.4 trillion. Without CH_4, the costs would be \$33.3 trillion.

8.4.5 CO_2 control revealed in referenda in Switzerland

A special case of analysing implicit values in policy decisions is the derivation of an implicit WTP for controlling CO_2 emissions from people's voting behaviour in referenda related to energy questions in Switzerland. Decision-making in Switzerland differs essentially from decision-making in other countries due to strong components of "direct democracy". In many cases, key Swiss policy issues are decided by a national referendum. There have been a number of Swiss national referenda related to the subjects "energy" and "environment". Some included decisions about prices/taxes. Referenda can be viewed as large surveys, which at the same time constitute political decisions.

The referenda provide the unique chance to study the revealed opinion of a huge number of people. The idea of the proposed method is to use results from referenda related to environmental issues to estimate preferences of the population.

Under plausible assumptions about the underlying WTP distribution, the average willingness of the Swiss population to pay energy taxes per kWh can be estimated. The referenda originally refer to taxes on non-renewable energy consumption in order to favour renewable energy. The change from fossil fuels to renewable energy affects mainly direct CO_2 emissions but not necessarily other pollutant emissions (e.g. NO_x or PM_{10}; emission factors for biomass are comparable to those for fossil fuels). Therefore it is plausible to account the WTP per kWh fully to CO_2 as far as emissions are concerned.

The resulting estimates are about €6 to 9/tCO_2 for the geometric mean and about €14 to 22/tCO_2 for the arithmetic mean. This estimate is of the same order of magnitude as the one derived from literature on cost-efficient implementation strategies to meet the Kyoto Protocol. The estimated WTP is however significantly lower than the abatement costs in Switzerland (starting at about CHF100/tCO_2, i.e. about €70/tCO_2).

8.5 Conclusions

The derived damage cost estimate is around \$33/tC = ca. €9/tCO_2 for a medium discount rate. However, this figure is conservative in the sense that only damage that can be estimated with a reasonable certainty is included; for instance impacts such as extended floods and more frequent hurricanes with higher energy density are not taken into account, as there is not enough information about the possible relationship between global warming and these impacts.

Thus, to account for the precautionary principle, we propose to use an avoidance costs approach for the central value. As discussed, the avoidance costs for reaching the broadly accepted Kyoto aim is roughly between €5 and €20 per t of CO_2. In addition it is now possible to look at the prices of the tradeable CO_2 permits, which increased from end of July 2005 to the beginning of October 2005 from about €18/tCO_2 to about €24/tCO_2. The large decrease in the beginning of September 2005 showed that the price still varies. This confirms the use of €19/t CO_2 as a central value. The lower bound is determined by the damage cost approach to about €9/t CO_2.

However, there is a tendency to strive for higher goals than the Kyoto ones. In addition, of course, estimates of the avoidance costs critically depend on the target chosen. The EU target of limiting global warming to 2°C above pre-industrial temperatures may lead to marginal abatement costs as high as \$350/tC = ca. €95/t CO_2. However it has still an open question whether such an ambitious goal with such high costs will be accepted by the general population. Thus, as an intermediate aim the Dutch value of ca. €50 per t could be used as an upper bound for a sensitivity analysis.

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9 Other Impacts: Assessment of Major Accidents

9.1 Major Accidents in Non-Nuclear Fuel Chains

In previous ExternE work before 2001, emphasis was placed on the quantification and valuation of impacts from *beyond design basis* accidents in the nuclear fuel cycle. However, other fuel chains also show a significant potential for severe accidents (e.g. oil fires or large spills, gas explosions, dam failures). The project NewExt (ExternE, 2004) reviewed and extended existing database systems on major accidents related to energy conversion activities. Furthermore, for hydro power an approach using elements of Probabilistic Safety Assessment (PSA) was defined and some of its components were elaborated on a limited-scope basis. In a second step, a methodology was developed to estimate external costs from major accidents, thus advancing comparability with the results earlier obtained for beyond design basis accidents in the nuclear fuel chain. This work allows for the first time a consistent and comprehensive assessment of externalities from major accidents in non-nuclear fuel chains.

9.1.1 Background

The main objectives of the work were (see Burgherr *et al*., 2004):

- To carry out comparative assessment of severe accidents in the energy sector, focusing on non-nuclear energy chains;
- To assess the external costs associated with severe accidents within the various energy chains.

Lack of estimates of external costs of non-nuclear accidents had previously been identified as one of the limitations of the state-of-the-art of externality assessment. The results obtained can support policy decisions and serve as an essential input to the evaluation of sustainability of specific energy systems.

In 1998 ENSAD (Energy-related Severe Accident Database), a comprehensive database on severe accidents with emphasis on the energy sector, was established by the Paul Scherrer Institute (PSI). The historical experience represented in this database was supplemented by probabilistic analyses for the nuclear energy, in order to carry out a detailed comparison of severe accident risks in the energy sector (Hirschberg *et al*., 1998). The database allows us to carry out comprehensive analyses of accident risks, that are not limited to power plants but cover full energy chains, including exploration, extraction, processing, storage, transport and waste management. The ENSAD database and the analysis have now been much extended, not only in terms of the data coverage but also the scope of data applications. For the full coverage of work performed we refer to Burgherr *et al*. (2004).

9.1.2 Database extensions and current status

The extensions of the ENSAD database and of the scope of analysis have taken place on various levels:

- Information from a variety of commercial and non-commercial data sources was added. Examples include specialised databases covering oil spills as well as dam accidents.
- The time period covered has been extended to reflect the historical experience to the year 2000 (previously it was 1996).
- Small accidents were also addressed although these accidents were not in the original scope of the study.
- Based on PSI's engagement in the China Energy Technology Programme of the Alliance for Global Sustainability, it has been possible to gain access to previously restricted information on accidents in China (Hirschberg *et al*., 2003a; Hirschberg *et al*., 2003b); records on Chinese accidents were practically unavailable in the past.
- Within the externality assessment, valuation of the relevant end-points (such as death and injury, evacuation of population, costs of oil spills) was carried out and the degree of internalisation was addressed.

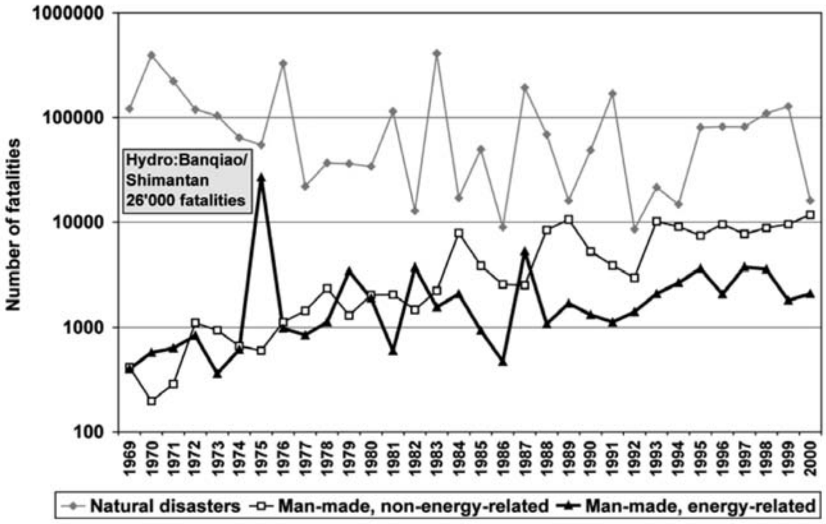

Figure 9.1 Number of fatalities in severe (≥ 5 fatalities) accidents that occurred in natural disasters and man-made accidents in the period 1969 to 2000.

In ENSAD an accident is considered as severe if any of the following seven criteria is satisfied: (1) At least 5 fatalities or (2) at least 10 injured or (3) at least 200 evacuees or (4) extensive ban on consumption of food or (5) releases of hydrocarbons exceeding

10,000 tons or (6) enforced clean-up of land and water over an area of at least 25 km^2 or (7) economic loss of at least \$5 million (in $US\$_{2000}$).

ENSAD currently contains 18,400 accidents. Man-made accidents comprise 12,943 or 70.3% of the total, whereas natural disasters amount to 5,457. A total of 6,404 energy-related accidents corresponds to 34.8% of all accidents or 49.5% of man-made accidents. Among the energy-related accidents 3,117 (48.7%) are severe, of which 2,078 have 5 or more fatalities. Non-energy-related accidents and natural disasters are of second priority within ENSAD. Consequently, the corresponding data are likely to be less complete and of lower quality than the ones provided for the energy-related accidents. Figure 9.1 shows the number of fatalities world-wide in different types of accidents over a period of more than 30 years.

9.1.3 Damage indicators and frequency-consequence curves

Selected aggregated accident indicators were generated and compared. The approach used accounts for contributions from all stages of the fuel cycles that were analysed. The comparison of different energy chains was based on normalised indicators combining consequences (e.g. number of fatalities) and product (e.g. electricity generation), and on the estimated accident-related external costs for selected technologies.

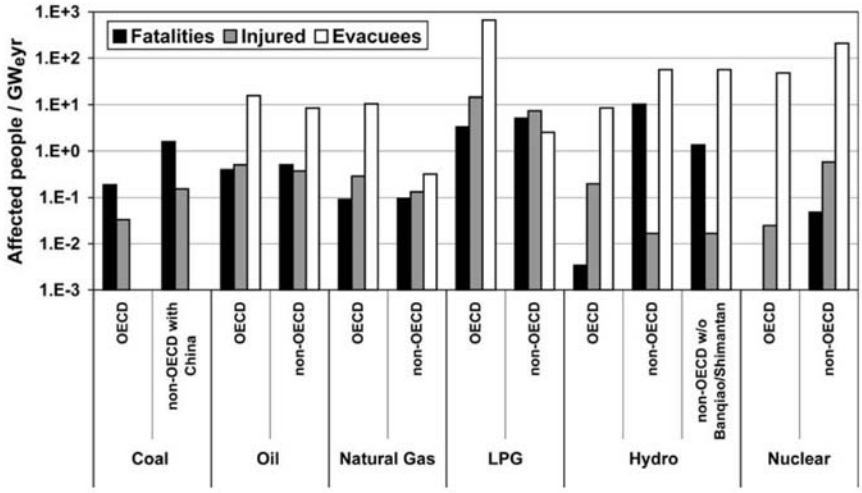

Figure 9.2 Aggregated damage rates, based on historical experience of severe accidents in OECD and non-OECD countries for the period 1969-2000. The indicators were estimated with partial reallocation of damages to OECD countries taking into account imports of fossil energy carriers from non-OECD countries. Only immediate fatalities are shown; latent fatalities will be commented on below.

Figure 9.2 shows results in terms of affected people per GW_eyr, differentiating between OECD and non-OECD countries[22]. It should be noted that the statistical basis for the indicators for individual energy chains may radically differ. For example, there are 1,221 severe accidents with fatalities in the coal chain and only one in the nuclear chain (Chernobyl).

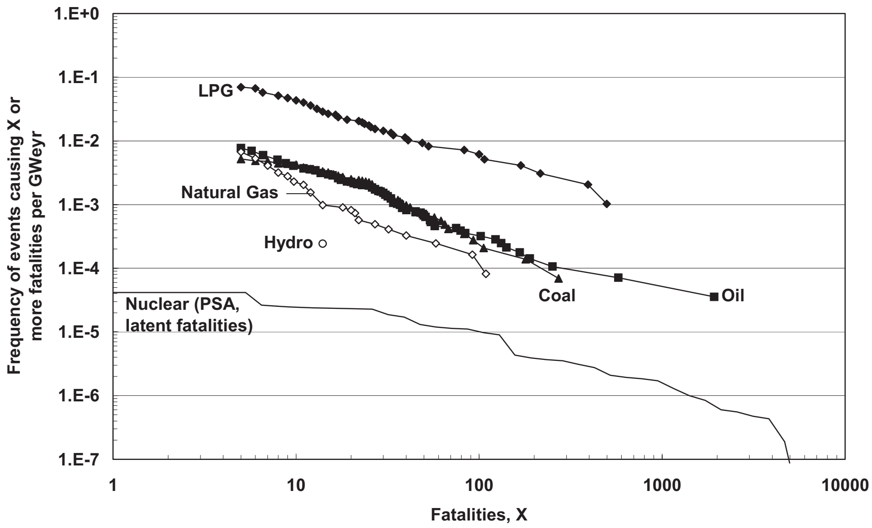

Figure 9.3 Comparison of frequency-consequence curves for full energy chains in OECD countries with partial reallocation for the period 1969-2000. The curves for coal, oil, natural gas, liquefied petroleum gas (LPG) and hydro are based on historical accidents and show immediate fatalities. For the nuclear chain, the results originate from the plant-specific Probabilistic Safety Assessment (PSA) for the Swiss nuclear power plant Muehleberg and reflect latent fatalities.

The frequency-consequence curves for OECD and non-OECD countries are provided in Figure 9.3 and Figure 9.4, respectively. Fossil-fuel energy chains in non-OECD countries display a similar ranking as for OECD countries, except for the Chinese coal chain, which exhibits significantly higher accident frequencies than in other non-OECD countries. However, the vast majority of severe coal accidents in China result in less than 100 fatalities. Accident frequencies of the oil and hydro chains are also much lower than for the (Chinese) coal chain, but maximum numbers of fatalities within the

[22] Corresponding results were also obtained for EU-15. With the exception of hydro power, they show no major differences compared to those for OECD countries. As the latter have a broader statistical basis they are also considered to be representative for EU-15. Specifically for hydro power there were no severe accidents in EU-15 during the period of observation.

oil and hydro chains are respectively one and two orders of magnitude higher than for the coal and natural gas chains.

Expectation values for severe accident fatality rates associated with the nuclear chain differ strongly between the two cases displayed in Figure 9.3 and Figure 9.4. The maximum credible consequences of nuclear accidents may be very large, i.e. in terms of fatalities comparable to the Banqiao & Shimantan dam accident that occurred in China in 1975. However, the large differences between Chernobyl-based historical estimates and probabilistic estimates for Muehleberg illustrate the limitations in the applicability of past accident data to cases which are radically different in terms of technology and operational environment. In this sense the Chernobyl accident is in fact also not representative for currently operating plants in non-OECD countries.

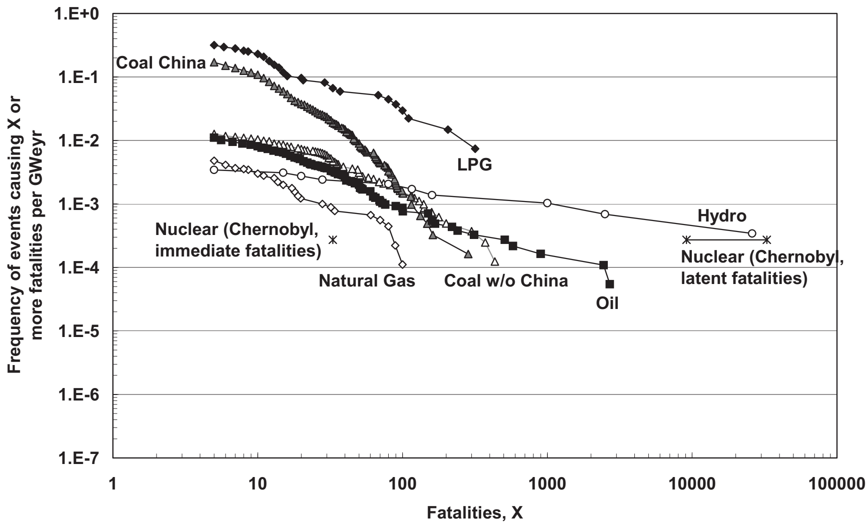

Figure 9.4 Comparison of frequency-consequence curves for full energy chains in non-OECD countries with partial reallocation for the period 1969-2000. The curves for coal w/o China, coal China, oil, natural gas, LPG and hydro are based on historical accidents and show <u>immediate</u> fatalities. For the nuclear chain, the <u>immediate</u> fatalities are represented by one point (Chernobyl); for the estimated Chernobyl-specific <u>latent</u> fatalities lower and upper bounds are given.

9.1.4 Conclusions

Damage costs and external costs of severe accidents in different energy chains can be estimated, based on the unit cost values for the various types of consequences as described in section 9.2.

- Comprehensive historical experience of energy-related severe accidents is available and can be used as a basis for quantifying the corresponding damages and external costs. Small accidents are strongly under-reported but their contribution to external costs appears to be quite small.
- Energy-related accident risks in non-OECD countries are distinctly higher than in OECD countries. The results obtained for OECD countries are also representative for EU-15.
- The results for OECD and non-OECD countries can be thus regarded as a lower and upper limit: For a European power plant that buys coal or oil from non-OECD countries, the attributable risks occur to certain shares within OECD but also non-OECD countries.
- Hydro-power in non-OECD countries and upstream stages within fossil energy chains are most accident-prone.
- Expected fatality rates are lowest for western hydropower and nuclear power plants. This results in low associated external costs. However, the maximum credible consequences are very large. The corresponding risk valuation is subject to stakeholder value judgments.
- The damages caused by severe accidents in the energy sector are substantial but quite small compared to those caused by natural disasters. External costs associated with severe accidents are quite insignificant when compared to the external costs of air pollution.
- Future comparative work on severe accidents should comprise: (a) Maintenance and further extensions accident databases; (b) Improvements of specific indicators (e.g. land contamination, economic damages); (c) Implementation of a simplified Probabilistic Safety Assessment for hydro power; (d) Use of a simplified state-of-the-art PSA-approach applied to several representative designs and European sites.

9.2 Valuation of Accidents in the Energy Supply Chain

The principal objective of this section is to derive unit values that express the welfare impacts of accidents in the non-nuclear energy supply chain in monetary terms, and enable calculation of the external costs of such accidents. Thus, for a given welfare impact unit, (e.g. a workplace injury), we look to identify a monetary value that represents the willingness to pay to avoid the impact or the willingness to accept (WTA) compensation to bear the injury. A taxonomy of external cost impacts that might result from a major accident in energy chains includes:

(i) Mortality (with or without hospitalisation) in accident;
(ii) Morbidity – physical injury in accident;
(iii) Mental trauma – from physical injury, evacuation;
(iv) Evacuation (costs of resettlement/accommodation);

(v) Clean-up/repair costs and willingness to pay (WTP) for recreational/ ecosystem losses – oil spills;
(vi) Ban on consumption of food;
(vii) Land contamination;
(viii) Other economic losses.

In the following sub-sections we summarise the possibilities for deriving appropriate unit values for each of these impacts. Our conclusions are drawn from the findings of a literature review that we have undertaken.

There is an established methodology – adopted in ExternE and related projects – for estimating the valuation of health risks. This involves – as the starting point for the valuation of health end-points and a number of the other impact categories considered below – the identification of the components of changes in welfare. These components should be summed to give the total welfare change, assuming no overlap between categories. As already mentioned in section 7.2.2, the three components include:

(i) Resource costs – medical costs paid by the health service in a given country or covered by insurance, and any other personal out-of-pocket expenses made by the individual (or family).
(ii) Opportunity costs – the cost in terms of lost productivity (work time loss (or performing at less than full capacity)) and the opportunity cost of leisure (leisure time loss) including non-paid work.
(iii) Disutility – other social and economic costs including any restrictions on or reduced enjoyment of desired leisure activities, discomfort or inconvenience (pain or suffering), anxiety about the future, and concern and inconvenience to family members and others.

We discuss the potential impact categories listed above with these components of WTP/WTA in mind.

9.2.1 Premature mortality

In the health economics literature, various methods for empirical estimation of willingness to pay measures have been utilised, each providing a method for deriving measures for individuals making trade-offs between risks to life and health and other consumption goods and services. These methods include the compensating or hedonic wage, the contingent valuation, the hedonic property value and the averting behaviour methods.

VPF measures in the energy supply accidents context
As described in previous ExternE documents, (e.g. European Commission, 1995a), estimation of the value of a lost life or of a prevented fatality (VPF) is fraught with

conceptual and empirical difficulties associated with the fact that there is no direct market for values to be reflected in. Two issues should be highlighted in relation to our present needs. First, estimates of the VPF that have been made to date have primarily been derived in the context of road traffic or workplace accidents. None is known to have been estimated in the context of energy supply operations and this therefore raises a question about the appropriateness of transfer between contexts. The second issue is that, in order to identify a unit value for the risk of premature death in the energy supply context, we need to consider whether or not – and to what degree – the WTP is measuring an external cost. For instance, if an employee who is working in the energy supply industry is fully compensated through the wage rate for the risk of a fatal accident to which he is exposed then the cost is fully internalised in existing financial flows. These two issues are discussed at some length below and we find it convenient to consider WTP for mortality risks to employees and the general public separately.

Work-related accidents

The derivation of a unit value for this impact is presented in two stages. First, we identify a VPF unit value, before estimating the extent to which the value is internalised in existing financial flows.

The hedonic wage method would seem to be the appropriate approach to empirically estimate work-related values of a statistical life, since it uses the wage-risk trade-offs (and other factors that affect wages) to estimate wage differentials related to different mortality risks. However, there are a number of difficulties associated with the estimation of VPFs using this method. Principal amongst these difficulties – based upon a review paper by Viscusi and Aldy (2003) – are the following:

- Risk data: the standard approach in the literature is to use industry-specific or occupation-specific risk measures reflecting an average of several years of observations for fatalities, which tend to be rare events. However, the choice of the measure of fatality risk can significantly influence the magnitude of the risk premium estimated through regression analysis.
- Omitted variables bias and endogeneity: failing to capture all of the determinants of a worker's wage in a hedonic wage equation may result in biased results if the unobserved variables are correlated with the observed variables, since dangerous jobs are often unpleasant in other respects. For example, one may find a correlation between injury risk and physical exertion required for a job or risk and environmental factors such as noise, heat, or odour. Various studies have demonstrated how omitting injury risk affects the estimation of mortality risk, indicating that a positive bias in the mortality risk measure is introduced when the wage equation omits injury risk.

  While including injury risk in a regression model could address concern about one omitted variable, other possible influences on wages that could be correlated with mortality risk may not be easily measured. For example, individuals may systematically differ in unobserved characteristics, which affect their productivity and earnings in dangerous jobs, and so these unobservables will affect their choice of job risk[23]. The studies reviewed by Viscusi and Aldy (2003) indicate that models that fail to account for heterogeneity in unobserved productivity may bias estimates of the risk premium by about 50%.

- Endogeneity: the issue here being that the dependent variable (wage) is explained by, among others, the risk variable, which simultaneously depends on wage, since "the level of risk that workers will be willing to undertake is negatively related to their wealth, assuming that safety is a normal good." Viscusi (1978). Gunderson and Hyatt (2001) empirically tested the alternative econometric models suggested by Viscusi (1978) and Garen (1988), identifying significant differences in the VPF estimates between the usual econometric model (OLS) and the proposed alternatives (€2.8 million to €12.8 million).

These difficulties with the reliability of the estimation methods are exacerbated when we try to identify a typical average unit value by the wide range of values that result from the wage-risk studies. A sample of the studies undertaken in the EU, presented in Table 7.1, section 7.2.1, demonstrates this.

As a consequence of the issues raised above, we do not find these estimates terribly robust. The alternative source of a unit value for a VPF is to use a value derived from other valuation methods. We recommend the use of the results of the valuation study described in section 7.2.1: €1 million for VPF and €50,000 for value of a life year lost (VLYL).

Fatalities that occur to employees involved in fuel cycles may already be at least partly internalised in producer costs, either through *ex ante* wages that account for fatality risks or through *ex post* compensation to families of the victim. Internalisation of the risk of fatality is likely to the extent that workers can be assumed to be well informed about the risks that they actually face in their work and that the part of the labour market to which these risks apply is competitive and flexible. Evidence of the validity of these assumptions is not easy to come by. In order to identify the degree to which internalisation of mortality and morbidity risks exists in the energy supply sector, we would ideally need to have a quantitative estimate of the extent to which actual wage rates differ from what they would be in a perfect market within this sector. There is no evidence from wage simulation models of this measure and results in this regard from wage-risk studies (the explanatory power of the risk variable) vary enormously.

In the absence of direct evidence of the degree of internalisation that we can assume, we have investigated the possibility of using a proxy for the degree to which workers

[23] Garen, J.E. (1988) "Compensating Wage Differentials and the Endogeneity of Job Riskiness", *Review of Economics and Statistics*, 73(4).

are well informed about mortality and morbidity, and are able to express this in wage negotiation and settlement. To this end, we have looked at the importance of education and unionisation as explanatory variables. Dorman (1996) finds that, whilst wage levels increase with the education level of labour force, there is no robust way in which these results can be related to differing levels of mortality/morbidity risk. Evidence regarding the role of unions (e.g. reported in CSERGE, 1999) in determining the level of risk premiums is also not particularly convincing since, whilst some studies found union affiliation had an insignificant impact on risk premium, others found that higher union risk premiums existed.

Given the lack of any satisfactory measure of internalisation, we are obliged to rely on judgement. On this basis we would suggest using 80% as a direct proxy value for the central degree of internalisation that may be assumed in OECD countries. High and low ranges of internalisation may reasonably be assumed to be 100% and 70% respectively, reflecting the fact that in industrialised economies occupational risk is recognised as being substantially internalised. For non-OECD countries we recognise that, whilst some economies, e.g. in Eastern Europe, are less effective and a lower degree of internalisation is to be expected, others, e.g. in East Asia, are much more market-orientated and are better able to reflect risk premiums according to the preferences of market participants. In the absence of hard data we suggest that a wide range of 0% to 100% internalisation, with a central value of 50%, is not too unreasonable to assume. It should be emphasised that the lack of data with which to validate these percentages significantly limits the extent to which they can be regarded as reliable.

Non-work-related accidents

In addition to work-related accidents, some fuel chain accidents affect a great number of people not related to the production *per se*, the general public. For example, floods generated from hydro-dam collapses may affect residents downstream of the dam. Two issues are important when considering valuation of risks of non-work related accidents in non-nuclear fuel chains: the fact that these risks are involuntarily taken by the population affected by accidents, and that the choices that individuals are able to make to allocate the perceived risks of potential accidents in fuel chains determine the degree to which the costs are internalised. These issues are considered in more depth below before making recommendations for final unit values.

The degree of involuntariness, or the lack of personal choice on the exposure to risks, may differ between different accident contexts. The argument here is that, whereas road accidents are more or less voluntary to the extent that the risk is in the individual's control and has responsibility for his/her actions, the degree of voluntariness can be judged to be very low for both employees and the general public who suffer fatalities from accidents in the fuel cycle. Evidence is sparse but one study (Jones-Lee and Loomes, 1995) identifies a 50% premium between the event of an underground train accident (involuntary) and road accident (voluntary), which did not appear to be the result of any particular additional dread of underground accidents relative to road accidents. It is proposed that this premium be adopted in sensitivity analysis.

Kunreuther (2001) argued that individuals can take two actions to reduce their losses from natural disasters and accidents and so internalise the risk: up-front expenditures to avoid or mitigate losses which provide benefits over the life, or the purchase of insurance which provides the policyholder with financial protection against a disaster loss for a fixed period of time in return for a premium to the insurance company. In determining which actions can be taken to reduce their losses from accidents, an individual would need to consider: the probability that the event (accident) will occur; the resulting loss associated with the event, and the cost associated with protection that reduces this loss from an accident. Normative models of choice predict that individuals, depending on their aversions to risk, maximise their utilities by choosing between two different protective measures, buying insurance or mitigation measures.

However, the empirical literature suggests that individuals and firms do not obtain the relevant data or do not undertake the (expected) utility maximising approach implied by normative models of choice. The factors that lead people to behave differently from what is predicted in normative models of choice are identified as (Kunreuther, 2001):

- Misperception of the risks – sometimes the probability of occurrence of a certain event is overestimated because of media coverage. For example, empirical tests suggest that the likelihood of deaths from widely reported disasters are perceived to be higher than those from events such as diabetes and breast cancer that are not reported in the media in the same way. Past experience may also play an important role in influencing individuals' perception of the probability of occurrence of an event. Individuals tend to perceive that an accident is more likely to occur after experiencing an accident than before the occurrence.
- Low probability events are perceived as impossible events – individuals tend to behave as if they consider the probability of the event occurring to be equal to zero, neither taking mitigating measures nor acquiring insurance.
- High discount rates – regarding investment in mitigating measures where the benefits are accrued over time, individuals may have a very high discount rate so that the future benefits are not given much weight when evaluating the protective measure.
- Imperfect capital markets – individuals may not have access to efficient capital markets and therefore may not be able to make a utility-maximising trade-off between accident risk and protection/compensation.
- Role of emotions – judgements on risks are based on dimensions other than probability and monetary losses, such as fear and dread, which have been shown to be very critical to individuals' risk perception. With regard to protective behaviour, studies found that people often buy warranties because they want to have peace of

mind or reduce their anxiety. In addition, presenting information to individuals in different ways may alter their perception of the risk.

- Ambiguity – or vagueness about the probabilities of losses related to given risks is an attribute, which seems to affect choices individuals make that is ignored in normative models of choice, such as expected utility theory. Empirical tests suggest that ambiguity in risks such as environmental pollution and earthquake losses does make a difference in individuals' willingness to pay to protect them against a risk.

As a consequence of this analysis, Kunreuther (2001) concludes that policies for dealing with low-probability-high-consequence events must consider a set of behavioural and capital market factors that are not considered in standard normative models of choice. It is also the case that insurance premiums in general cover only the material losses from loss of income, for example, and not the costs imposed by pain, suffering and trauma.

With these issues in mind, we have reviewed the level of insurance compensation payments that are made in the EU. *Ex post* evidence[24] suggests that liability insurers pay a mixture of lump sum and annuities related to wage losses and medical costs for injuries (although in France the indexation is borne by the state) and a mixture of lump sum and annuities related to wage losses to the family for fatalities.

Coverage for accidents varies over countries and industries but, on average, between 70% and 80% of material losses are paid, i.e. internalised. We therefore assume that 75% of material losses are paid. In order to account for pain and suffering that is not included in standard compensation payments, we make a conservative assumption that this component is equal to 50% of the value of the true material losses. Thus, with a compensation payment made of €500,000, these assumptions imply a full material cost of €666,666 (1/0.75 · 500,000) and a full WTP value of €1 million (adding in 50% of €666,666), showing that the compensation payment made is 50% of the full internalisation for OECD countries. We adopt this as a central value, with a range of between 30% and 70%. For non-OECD countries, we suggest that a range of between 0% and 50%, with a central value of 20% would be reasonable. Again, the evidence to support these ranges is weak but is based purely on the knowledge that many of these countries are characterised as having imperfectly functioning market economies.

While this approach allows for the internalisation of some of the risk, we should note that the component that is internalised is also of interest to policy-makers. It reflects the shifting of the costs of using a resource from the producer of energy to the general public. Hence we recommend that the internalised values also be reported alongside the externalities.

[24] http://www.munichre.com/pdf/claims_life_and_ancillary_benefits_e.pdf

Final Remarks on value of a prevented fatality

There are three further factors that have been hypothesised as influencing the individual's valuation of a risk of death from fuel cycle related accidents. We discuss these in the following paragraphs before providing a summary table of recommended values.

<u>The scale of the accident</u>

It has been hypothesised (Savage, 1993) that the scale of an accident (in terms of number of fatalities resulting) may influence the WTP valuation of accident fatalities i.e. that risks of large-scale accidents may be valued more highly. There is to date little evidence available to test this hypothesis. However, a study by Jones-Lee and Loomes (1995) compares the valuations that arise out of WTP for large-scale underground train accidents and third party accidents from proximity to airports with those from small-scale road transport accidents. They found no evidence of a significant scale premium, apparently reflecting in part, people's doubts about the preventability of rare, large-scale accidents and the consequent reservations concerning the effectiveness of expenditure aimed at their prevention.

<u>Non-linearity of the size of risk (probability of accident)</u>

It has been noted in earlier ExternE projects that the probability range over which the valuation of mortality risk has been undertaken in road accident studies is typically 10^{-1} to 10^{-5}, whereas the probability of death from accidents may be more likely to be of the order of 10^{-6}. Furthermore, it has been suggested (Lindberg, 1999) that values of mortality risk vary in a non-linear way. As noted in earlier ExternE reports, the evidence is not currently sufficient to make any firm proposals on such an adjustment at present.

<u>Spatial transfer of unit values</u>

Given that incidents of mortality from the non-nuclear fuel cycle occur globally there remains a question as to the appropriate basis for transferring values between EU countries and outside the EU. We recommend that, for EU countries themselves, there should be no differentiation between individual countries and that common EU values should be utilised. For mortality incidents that occur outside of the EU, economic theory suggests that, from an efficiency perspective, – if income is assumed to be the principal variable in explaining cross-regional variation – the values could be disaggregated on the basis of local resource costs. In practice, this is measured by purchasing power parity (PPP), and this ratio – referenced to the EU15 – is what we recommend for use here[25]. The PPP ratio should be used for individual countries in future policy analysis. However, where the spread of countries impacted is not known, we recommend the use of the unadjusted EU unit value.

[25] Transfer functions sometimes consider differentials in income elasticities between countries or regions. For a detailed discussion, refer to Markandya (1998) "The Valuation of Health Impacts in Developing Countries." *Planejamento and Políticas Públicas*, n.18, Dec.

In Table 9.1 we present the unit values that should be used in quantification of accident-related mortality impacts in OECD and non-OECD countries. We assume that the central non-OECD country estimates are representative of industrialised countries of similar per capita income levels to those prevailing in the EU. The minimum and maximum ranges reflect the considerable uncertainty that remains in the derivation of these values. It is recommended that these ranges be used in all quantification of mortality impacts in policy analysis. In addition, the 50% premium of involuntariness exposure to risk noted above should be included in further sensitivity analysis.

Table 9.1 External mortality unit values (€ 2002).

	Central	Minimum	Maximum	Proportion of internalisation
Value of a Prevented Fatality	1,000,000	400,000	3,310,000	
Occupational fatalities				
Central OECD (80%)	200,000	80,000	662,000	0.8
Lower internal OECD (70%)	300,000	120,000	993,000	0.7
Upper internal OECD (100%)	0	0	0	1.0
Central Non-OECD (50%)	500,000	200,000	1,655,000	0.5
Lower internal Non-OECD (0%)	1,000,000	400,000	3,310,000	0.0
Upper internal Non-OECD (100%)	0	0	0	1.0
Public fatalities				
Central OECD (50%)	500,000	200,000	1,655,000	0.5
Lower internal OECD (30%)	700,000	280,000	2,317,000	0.3
Upper internal OECD (70%)	300,000	120,000	993,000	0.7
Central Non-OECD (20%)	800,000	320,000	2,648,000	0.2
Lower internal Non-OECD (0%)	1,000,000	400,000	3,310,000	0.0
Upper internal Non-OECD (50%)	500,000	200,000	1,655,000	0.5

Note: Percentages refer to the degree of risk internalisation

9.2.2 Morbidity

Much of the discussion that applies to valuation of mortality risks from accidents applies to the valuation of injuries. Unfortunately there is not a single study on which we can rely to provide us with baseline unit values. Therefore, we rely on the work of Lindberg (1999) who usefully summarises the ratios between fatality values and values for severe[26] and minor[27] injuries. He concludes that the recommendation made by

[26] Severe injuries include amputation, major fractures, serious eye injuries, loss of consciousness and any injuries requiring hospital treatment over 24 hours

ECMT (1998), of weighting the risk value for severe injuries at 13% and for minor injuries at 1% of the risk value of fatalities, is broadly supported by the evidence from individual, generally CVM, studies – although the studies reflect a wide range of values. These ratios – and the unit values they generate – are also consistent with injury values adopted in previous ExternE work. The unit values of injuries are reported in Table 9.2. However, whilst Lindberg splits injuries into “severe” and “minor” categories, the historical data on incidence of injuries resulting from fuel cycle accidents does not disaggregate in this way. Consequently, the bottom line in the table presents unit values for a “typical” injury, represented by the mean of the “severe” and “minor” categories.

Table 9.2 Morbidity unit values (€ 2002).

	Central	Minimum	Maximum
Value of a Prevented Fatality	1,000,000	400,000	3,310,000
Severe injury	130,000	52,000	430,300
Minor injury	10,000	4,000	33,100
"Typical injury"	70,000	28,000	231,700

9.2.3 Mental trauma

It is recognised that the mental trauma of being impacted by fuel cycle related accidents might be a significant welfare effect in some instances. For example, should there be a hydro-electric dam breach in a given area, it is likely to affect those who live close by directly, by requiring them to move, or by flood damage, or indirectly because of their proximity and perceived vulnerability. Another example may be the trauma that follows from an oil platform accident that injures or kills other colleagues. There are therefore public and occupational valuation issues that need to be considered in this context.

The principal difficulty with deriving monetary values for this impact category is that it is intangible and has psychological effects that cannot easily be identified or quantified in any meaningful way. It is therefore difficult to rank the severity of mental trauma experiences and differentiate among them in monetary terms. This difficulty is combined with the fact that mental trauma is often experienced concurrently with a physical effect e.g. injury or evacuation. To some extent, it would appear possible in the case of physical injury that mental trauma is being picked up in the valuation of the disutility component. In the context of evacuation or proximity to a severe accident, this is not so. One methodological possibility for valuing mental trauma is to multiply our mortality range values by a fraction determined by disability weightings that accord with individual mental health conditions. For example, the Dutch Disability Weights project gives a weighting of 0.76 of a life year lost to the condition of severe

[27] Minor injuries include other accidents responsible for the loss of more than three working days

depression[28]. However, there is no information available on the length of time associated with the mental trauma end-point. As a rough guide we suggest using a value of one year as reported since this is regarded as typical for flood damage victims.

9.2.4 Evacuation / resettlement

Severe non-nuclear fuel cycle accidents such as hydroelectric dam failure and gas/oil leaks/spills have led to the temporary or permanent displacement of people from their homes and/or places of work. This clearly has welfare impacts and these might include tangible costs including damage to property and other economic assets, transport, food and accommodation costs, medical and miscellaneous costs, and subsequent income losses. Some of these costs (e.g. property, medical and employment) may have been internalised to the extent that private insurance payments cover these events. Intangible costs relate to disutility and may include mental trauma of the type noted above.

A survey of the literature has provided estimates of evacuation costs from the USA, but not for the non-nuclear fuel cycle. Two studies, one from the context of a simulated radioactive evacuation, the other from the hurricane evacuation context, have estimated unit values. The first (Radioactive Waste Management Associates, 2000) makes estimates of direct economic costs using two categories: fixed evacuation costs of €180 per family. The second (Tyndall Smith, 2000) gives the following mean approximate total costs of evacuation per household: €25 for accommodation, €50 food, €25 travel, €3 entertainment and €5 miscellaneous, summing up to €108. No medical costs are included in this latter study. On the basis of this evidence, we use the transfer value range of €108 to €180 for fixed direct economic costs with a mid-point of €144.

There will also be the loss of output resulting from absenteeism for work over the length of evacuation period. A survey study in the UK (CBI, 1998) has calculated the direct cost of absence, based on the salary costs of absent individuals, replacement costs (i.e. the employment of temporary staff or additional overtime) and lost service or production time. This amounts to €88/day absence. We note, however, that indirect costs of absenteeism (i.e. costs relating to lower customer satisfaction and poorer quality of products or services leading to a loss of future business) are not included. The UK survey estimates that these are €160/day absence, although this value was based on a small sample size. Including both elements produces a total of €248/day absence – we suggest that this should be the maximum value in a range from €88/day absence. A mid-point of €168 is a central estimate. There is no estimate available for the disutility of suffering evacuation although this might be thought to be very

[28] Stouthard MEA, Essink-Bot ML, Bonsel GJ, Barendregt JJ, Kramers PGN, Van de Water HPA, Gunning-Schepers LJ, Van der Maas PJ. Disability Weights for Diseases in The Netherlands. Rotterdam, The Netherlands: Department of Public Health, Erasmus University, 1997.

substantial. Clearly, there is overlap with the discussion of mental trauma – for which, as noted above, WTP values are elusive.

Resettlement costs associated with the construction of dams are presented in Table 9.3, although these are all in relation to countries outside the EU. Comparison, however, is limited by inconsistency with regard to the cost elements included in estimates for individual dams. For this reason, robust unit values are difficult to recommend and we therefore do not make any recommendations for this impact end-point.

Table 9.3 Resettlement costs from construction of dams (€ 2002).

	Construction	Resettled	Resettlement €	Cost per person €
James Bay, Canada	1995	18,000	594,940,000	33,052
Akosombo, Ghana	1965	80,000	50,000,000	625
Theun Hinboum, Laos	1998	25,000	2,600,000	104
Iron Gate 1, Romania	1971	24,000	69,300,000	2,888
Pak Mun, Thailand	1994	4,945	23,000,000	4,651
Kariba, Zimbabwe	1959	57,000	601,000	11
Nam Ngum, Laos	1972	3,474	58,500,000	16,839
Lesotho Highlands WP	2017	8,400	43,000,000	5,119
Magat, Philippines	1983	2,150	8,214,285	3,821
Kotomale, Sri Lanka	1985	13,000	4,251,249	327
Hunan Lingjintan, China	1996	4,275	28,140,678	6,583
Shuikou, China	1993	84,400	209,547,000	2,483
Average Non-OECD				**3,950**

Source: Input to World Dam Database, World Commission on Dams

As with the welfare impacts of evacuation, these cost estimates do not include estimates for disutility. These could, in theory, be estimated using either contingent valuation or hedonic price techniques. We are not aware of any such estimates being made for this impact. We suggest that the unit values for evacuation should be adjusted by PPP for non-OECD countries in policy analysis. In the absence of specific country contexts, it seems most sensible to use the un-adjusted values given here.

9.2.5 Ban on consumption of food

We might expect a welfare impact to result from changes in food commodity prices and quantities as a result of a ban on food consumption following a contamination incident. Such a ban on consumption could be expected as a result of oil spills both on land and/or in aquatic biomes. Empirical estimates from the non-nuclear accident context are not easy to come by – indeed it is unlikely that estimates, were they available, would be transferable since there is likely to be a high degree of context specificity.

However, whilst not related to the non-nuclear fuel chain, the compensation to farmers on beef ban in UK, presented in Box 1 below, provides an illustration of the producer surplus element of the associated welfare loss. This could be used as an indicator of the magnitude of costs involved in bans on consumption of food.

Box 9.1 Cost of beef ban in the UK

In the Spring of 2001 the spread of foot and mouth disease in the UK led to the statutory precautionary slaughter of any cow and sheep herds which either contained diseased animals or which – through their location – might have been carrying the disease. In April 2001, the British government announced a scheme for compensating beef, dairy and sheep farmers affected by the foot and mouth disease. Farmers received full market value for slaughtered animals. In addition, compensation was paid for any feeding stuffs or any other materials destroyed or seized as being possibly contaminated, which could not be satisfactorily disinfected.

The compensation scheme, approved by the European Commission on 3rd April 2001, involved payments of €30 per head of cattle and €2.2 per sheep. This gave a total of €180 million initially and a further €35 million for the beef sector in Autumn 2001 – equivalent to just under five percent of the total UK sectoral output.

9.2.6 Land contamination

Costs of restoring land to the condition it was in before a fuel cycle accident can be estimated from existing experience of clean-up of areas that have been contaminated by similar substances that are likely to contaminate from fuel cycle accidents. Of course it should be remembered that cost estimates such as these based on actual expenditures made represent minimum estimates of WTP values. WTP values may, however, be derived from the economic values that accrue to the owners of the land once the land is restored and put to economic use, above what they would have been in its contaminated condition. We have not been able to make assessments of appropriate unit values because of the lack of available data. Future work would – in any case – be best undertaken in specific contexts since this impact category does not lend itself to generic transfer of values.

9.2.7 Economic losses

Economic losses are likely to result from severe accidents in addition to those identified in the categories above if business operations are disrupted for example. In principle, economic losses can be estimated by changes in market supply and demand conditions in a partial equilibrium welfare analysis. As an example of estimates of economic losses due to oil spills we note a study conducted by Cohen (1995) who employed a market model to evaluate the economic losses of the 1989 Exxon Valdez oil spill on

Alaska's fisheries. The methodology used involved a three phase *ex post* forecasting approach to estimate economic losses from the oil spill. First, the author estimated provisional values of the accident's harvest volume impacts in each of the fisheries affected. Second, initial estimates were derived of the ex-vessel prices of regionally harvested fish and shellfish that would have prevailed in the absence of the oil spill. Finally, the (econometric) analysis constructed several alternative simulations to isolate the accident's social costs from a number of confounding biological and economic factors.

Determination of the social costs of the Exxon Valdez oil spill on Alaska's fisheries involved estimating the difference between the economic benefits that would have been derived in the absence of the oil spill with those derived in the presence of the accident. The social costs of the oil spill on Alaska's fisheries during 1989, based on the provisional estimates of the accident's harvest volume and ex-vessel price impacts, were US$108.1 million. In 1990, the oil spill's social costs on Alaska's fisheries were estimated to have been US$47.0 million. As with land contamination impacts, we do not recommend the transfer of unit values based on these, or other, estimates due to the highly context-specific nature of such incidents.

9.2.8 Clean-up/repair costs and willingness to pay (WTP) for recreational/ ecosystem losses – oil spills costs

The welfare impacts of oil spills are likely to be determined by the scale of the spill, the ecological services that the impacted area supports and the scale and nature of "human" related services affected in the area. Estimation of these welfare impacts has had a certain level of attention in the wake of a number of high profile oil spills – primarily in the Atlantic and North Sea regions. In theory, welfare valuation should be estimated by calculating the different components of total economic value: direct and indirect/passive use plus non-use values. Economic assessments have been undertaken; the results for two are summarised below.

1996 Sea Empress oil spill – Atlantic, off the South Wales coast, UK
Approximately 72,000 tonnes of crude oil and 480 tonnes of heavy fuel oil were released into the sea, and 100 km of coastline were affected. Commercial and recreational fishing was banned for seven months and the tourism industry was affected. Large numbers of marine organisms were killed as well as several thousand sea birds. The financial and economic costs are summarised in Table 9.4 below.

Note that the lower and upper bounds reflect the uncertainty as how to ascribe best measures of costs to the oil spill. Note also that the economic costs are greater than the financial costs for the conservation of ecosystems and their non-use values, reflecting the fact that these costs – whilst having welfare effects – are not reflected in financial market prices.

Table 9.4 Summary of total costs resulting from Sea Empress oil spill (£m).

	Financial costs		Economic costs	
Category	Lower bound	Upper bound	Lower bound	Upper bound
Direct clean-up costs	49.1	58.1	49.1	58.1
Tourism	4	46	0	2.9
Recreation	-	-	1.0	2.8
Commercial fisheries	6.8	10	0.8	1.2
Recreational fisheries	0.1	0.1	0.8	2.7
Local industry	0	0	0	0
Conservation/non-use	-	-	22.5	35.4
Human health	-	-	1.2	3
Total	60.0	114.3	75.3	106.1

Source: Environment Agency, "Sea Empress Cost-Benefit Project". HMSO (1998).

1989 Exxon Valdez Oil Spill, Gulf of Alaska
Approximately 39,000 metric tonnes of crude oil was released in Prince William Sound, before spreading to the Gulf of Alaska and 1,300 miles of coastline were oiled. There were acute damages to seabirds (250,000 dead), bald eagles, marine mammals and inter-tidal communities. Longer-term impacts were borne by Pacific herring, pink salmon and the inter-tidal and sub-tidal environments. Assessments of the impacts varied between scientists a decade after the event. The most detailed estimates of welfare impacts that exist derive from the compensation payments made by Exxon as a result of combined civil and criminal settlements. These payments included the following:

- Civil settlements
  - WTP damage assessment (including passive use values, aesthetic and non-use measured by CVM), litigation and clean up: €213 million;
  - research, monitoring and general restoration: €180 million;
  - habitat protection: €395 million;
  - long-term restoration: €108 million;
  - science management, public information and administration €31 million;
- Criminal settlements
  - habitat protection and improvements: €100 million;
- Total economic damage equated to €1.027 billion.

In order to derive unit damage values for future damage risk assessment, we can derive damage cost per tonne of oil in the two examples. This produces values of €26,333 and €2,368 per tonne of crude oil for Exxon Valdez and Sea Empress, respectively. The difference can be explained partly by the fact that different elements of total economic value were given attention in the two cases, partly by the fact that the damage in the case of oil spills is clearly contingent upon location and weather conditions at the time that determine dispersal patterns, and, of course, partly by different preferences between populations. For these reasons the most sensible course of action in making recommendations of unit values is to suggest a range of unit values that could be used in risk assessment exercises that might inform policy. The lower value, derived from the Sea Empress incident, is in fact supported by evidence from a number of oil spills in the Caspian Sea that have resulted in average damage costs of €2,600 per tonne. We therefore take this modal average as a central value. As a consequence we suggest that the best indicative unit values to use are:

- Central: €2,600/tonne
- Minimum: €2,300/tonne
- Maximum: €24,000/tonne

These are clearly not robust values to be relied upon in all contexts and we would not make any differentiation between OECD and non-OECD countries. Nevertheless these values provide a useful range with which to work.

9.3 Major Accidents in the Nuclear Fuel Chain

9.3.1 Background

The methodology for the damage costs of the nuclear fuel chain was developed by CEPN in France and initial results were published in 1995 (European Commission, 1995b). The damage costs were then calculated for five representative power plants in France (Rabl *et al.,* 1996). Using essentially the same methodology, the damage costs were estimated for the nuclear fuel cycle in Belgium, Germany, the Netherlands and the UK during the National Implementation phase of ExternE (European Commission, 1999). The results are of the same order of magnitude as those in France, but somewhat different, mainly for the following reasons:

- The technologies and emission factors are different, in particular with regard to radon emissions from uranium mines after their closure;
- Significant global warming damages have been added because the electricity for the isotope separation stage has been assumed to come from the current power plant mix in the respective countries rather than from nuclear. This allocation, made by some assessments, is not appropriate because isotope separation is needed only for nuclear power and it requires baseload electricity as provided by nuclear. Therefore the logically correct allocation of this electricity is to nuclear, with negligible emission of greenhouse gases.

Since then no further work has been done on the damage costs of nuclear power. That is most regrettable because the methodology could and should be improved, and it should be applied to current and future technologies which are safer and cleaner than the ones of the mid 1990s that have been considered until now.

Radiological impacts from emissions during power plant operation and final disposal were found to be only of minor importance for the overall results from the nuclear fuel cycle. For this reason, and as there was no improvement in the methodology to date, we refer to European Commission (1999). As potential damages from nuclear accidents play an important role in the public perception of risks from nuclear power production, the following section reproduces the key issues in the assessment of accidents in the nuclear fuel chain.

The methodology used to evaluate impacts due to accidental releases was risk-based expected damages. Risk is defined as the summation of the probability of the occurrence of a scenario (P_i) leading to an accident multiplied by the consequences resulting from that accident (C_i) over all possible scenarios. This can be simply represented by the following equation:

$$\text{Risk} = \sum P_i \cdot C_i \qquad (9.1)$$

9.3.2 Severe reactor accidents

A comprehensive probabilistic safety assessment (PSA) of potential reactor accidents did not fall within the scope of the project phases under consideration. In addition, the detailed data on potential source terms and associated probabilities for a multitude of potential scenarios for nuclear power plants were not available. For France four hypothetical scenarios were evaluated in order to demonstrate the range of results 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 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; so in the study considered 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 is assumed that the probability of massive containment failure or bypass conditional upon a core melt is 0.19 and the probability of the containment remaining intact is 0.81 (NRC, 1990). The same assumptions were made for the 900 MWe PWR assessment (Dreicer *et al.,* 1995).

The reference scenario ST21 for France assumes a containment failure that results in the total release of about 1% of the core on the average (10% of noble gases from the core, 1% of the more volatile elements, such as caesium and iodine, and smaller percentages of other elements). This source term is of the same order of magnitude as the reference accident scenario considered by the French national safety authorities (S3) (Queniart *et al.,* 1994). The three other source terms evaluated to illustrate the sensitivity of the results are respectively 10 times larger than ST21 (ST2, massive containment failure with 10% of the core released) and 10 and 100 times lower than ST21 (ST22 and ST23).

The public health impacts and economic consequences of the releases were estimated using the EC 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 must be altered, as is shown in Figure 9.5. The priority atmospheric release pathways for local and regional areas out to 3,000 km from the site were assessed. The definition of time and space boundaries are not the same as those defined in the assessment of routine operations of the fuel cycle.

The monetary valuation of the health effects arising from the collective dose is completed in the same manner for all the other parts of 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.

The use of this type of methodology does not necessarily include all the social costs that might result after a severe nuclear accident. One important issue is the social costs of risk aversion. Further work is required before the external costs of a severe accident can be considered complete.

For the analysis of nuclear accidents from a PWR reactor, different source terms for release have been used as base data in France, Germany and the UK. 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 presented in Table 9.5.

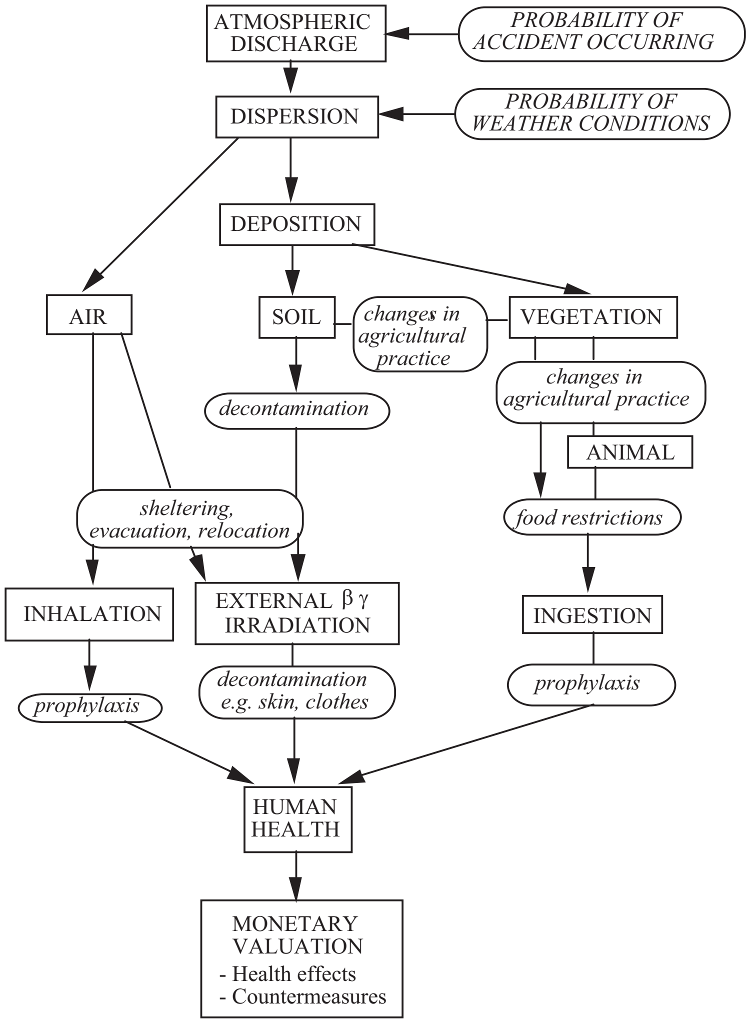

Figure 9.5 Pathways for a severe accidental release.

This variation makes cross-country comparisons difficult. 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.

Table 9.5 Release scenarios for different countries (Markandya *et al*., 1998).

Country[1)]	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 of Cs
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

[1)] The country results of France and Germany refer to the ExternE National Implementation study. For UK, the earlier Hinckley Point study (NRPB 1988) was examined. The National Implementation study in UK focused on another pressurised water reactor located at Sizewell but did not quantify the effects of major accidents.

The results on beyond design accidents for Germany (achieved in the National Implementation phase of ExternE – see European Commission, 1999) amount to €-cent 0.00034-0.00046/kWh (at that time expressed in "European Currency Units" – ECU) using a discount rate of 0 % and €-cent 0.000050 – 0.000076/kWh using a discount rate of 3 %. The lower value results from using the value of a life year (VOLY), the higher one from the value of a prevented fatality (VPF).

In spite of different assumptions on probabilities and source terms, the results for France are similar to those for Germany. In the reference scenario, representing a core melt accident followed by a release of 1 % of the core, external costs of accidental situations resulted in €-cent 0.0005/kWh using the VPF valuation and a discount rate of 0 % (Spadaro and Rabl, 1998). In total, the share of severe accidents in total external costs of the nuclear fuel cycle was found to be negligible for France and Germany.

A summary of key outcomes for the French nuclear fuel cycle has also been presented by Schieber and Schneider (2002). Trying to take "risk aversion" into account leads to an increase of the evaluation of a nuclear accident by a factor of 20. A newer overview of external cost studies for nuclear electricity generation is shown by NEA/OECD (2003); however this does not lead to newer insights. Even with the inclusion of indirect effects and a risk aversion factor of 20, the estimated cost of a nuclear accident represents less than 5 % of the external costs of the nuclear fuel cycle. However, the Nuclear Energy Agency highlights the need for further work on methodologies and tools to evaluate the impacts of accidents and their monetary values (NEA/OECD, 2000).

It is sometimes argued that, for so-called *Damocles risks*, i.e. risks with a very high damage and a low probability, the risk assessment of the public is not proportional to the risk. The occurrence of a very high damage should be avoided, even if the costs for the avoidance are much higher than the expectation value of the damage. However past attempts to quantify this effect have not been successful or accepted, so there is currently no accepted method on how to include risk aversion in such an analysis. Consequently it is currently not taken into account within the ExternE methodology. Research on how to assess this, for example with participatory approaches, is clearly needed.

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10 Other Impacts: Ecosystems and Biodiversity

The valuation of impacts on ecosystems and biodiversity is an example where the impact pathway approach cannot be applied fully, because quantitative knowledge on physical impacts and data on valuation are limited.

In the context of valuing biodiversity a wide range of contingent valuation studies have been undertaken, which either focused on valuing endangered or rare species or biodiversity programmes for specific local landscapes/habitats. In the NEEDS project a new methodology for assessing biodiversity losses due to energy production is being developed: changes in biodiversity resulting from emissions of SO_2 and NO_x and land use changes are measured for different habitat types. The resulting biodiversity changes are then monetised using a restoration cost approach. This approach will be documented in future reports.

In the following section a second best approach for valuing the impacts of acidification and eutrophication on ecosystems (based on the critical loads concept) is presented.

10.1 Acidification and Eutrophication

Acidification is mainly caused by emissions of sulphur dioxide (SO_2), nitrogen oxides (NO_x) and ammonia (NH_3), while eutrophication by airborne pollutants is mainly caused by NO_x and NH_3. Evidence has shown that acidification has a potential negative effect on aquatic and terrestrial ecosystems, surface water, agricultural and forestry yields, buildings and human health. Also eutrophication, or the enrichment by nitrogen nutrients of soil, ground water and surface water, results in a potential negative effect on aquatic and terrestrial ecosystems, surface water, and agricultural and forestry yields.

Due to missing quantitative dose-response relationships which would allow the calculation of impacts on ecosystems, the accounting framework has to be extended to include the environmental impact categories which are the main driving force for some of the most important international energy and environmental policy actions (EU acidification strategy, EU NEC directive, UN-ECE LRTAP Protocols).

The following sections presents the approach and application for evaluation of environmental impacts based on preferences revealed in political negotiations.

10.2 Methodology

The *standard price approach*, an approach based on the implicit values of policy makers, estimates the revealed preferences of policy makers. It calculates the benefits of emission reduction – as perceived by policy makers – based on the abatement costs to reach a well-defined emission reduction target. These costs are a proxy for the benefits that policy makers attribute to these reductions, as we assume that policy makers act as rational decision makers who carefully balance (their perception of) abatement costs of emission reductions with (their perception of) the benefits of these emissions.

As the *standard price* approach is based on the current preferences of policy makers, as reflected in air quality policies, it cannot be used for cost-benefit analysis or policy advice related to these emission reduction policies. Nevertheless, this *second-best* method gives useful data for comparison of energy technology and fuels because it gives us 'shadow prices' for a non-market scarcity, i.e. protected ecosystems from acidification and eutrophication.

10.2.1 Estimating the shadow prices per tonne pollutant for impacts on ecosystems

Earlier studies have used abatement costs as 'shadow prices' for the total impacts on human health, agriculture and ecosystems, expressed as € per tonne pollutant. We follow a more sophisticated approach, which aims at figures that are more in line with the *impact pathway* approach of ExternE and that are additive to the ExternE estimates for impacts on human health, agriculture and building materials. Therefore, the analysis combines the impact pathway approach to estimate impacts in physical terms (step 1), which are then valued following a careful analysis of international agreements of emission reductions in Europe (step 2). On this basis, we can estimate the shadow price per tonne of emissions (step 3).

10.2.2 Selection of the emission reduction programmes and determination of the WTP

The implicit values of policy makers associated with the protection of ecosystems have been defined in terms of pushing back (closing the gap) the number of hectares of ecosystem that remain unprotected from acidification and eutrophication. Therefore, in step 2 of the analysis, we have to determine the society's WTP for 1 hectare of ecosystem protected. The calculations of step 2 result in one figure for the whole of EU-15 for each emission reduction programme examined. The different assumptions made and parameters used in the analysis are as follows:

We have to define a marginal cost curve for emission reductions and select an emission reduction level that has been agreed upon by the policy makers. To this purpose we have analysed two emission reduction programmes and a reference scenario for the year 2010:

- Reference scenario (REF): The 'reference' scenario can be seen as a target for 2010 based on a 'business as usual' scenario starting for the status in 1998 (Amann *et al*., 1999a).
- Protocol of Gothenburg on the Convention on Long-range Transboundary Air Pollution (1999) (PRO): The policy goals of PRO are based on making significant progress towards reaching a scientifically based objective, i.e. a reduction by 50% of the number of hectares of ecosystems facing an exceedance of their 'critical loads' for eutrophication and acidification for the year 2010 (UN-ECE, 1999).
- European directive 2001/81/EC on National Emission Ceilings for some air pollutants (NEC): The original proposal on national emission ceilings of the European Commission (NEC+) of 9 June 1999 was much more ambitious and scientifically better grounded than the PRO. The directive finally adopted in 2001 (NEC) was set less ambitiously so that the emission levels are only slightly stricter for NEC than for PRO. NEC has the same policy targets as PRO (European Commission, 2001).
- PRO and NEC are both based on a 'multi-source, multi-effect approach', taking into account a multitude of sources and locations of emissions and a multitude of receptors and locations for deposition. The policy does not only focus on the effects of acidification and eutrophication by SO_2, NO_x and NH_3 on ecosystems, but also those of ground-level ozone from NO_x and VOC emissions on human health, agriculture and ecosystems.

Basic assumptions made in the analysis

- The number of hectares of ecosystem for which critical loads for acidification and eutrophication have been exceeded has been used as the physical indicator to evaluate the effects of acidification and eutrophication on ecosystems. This study does not question the use of the critical loads approach as a physical indicator. Although this is in line with the indicators used in a wide range of scientific and policy documents, it does not fully reflect all marginal impacts on all ecosystems.
- We simply add up exceedance for different types of ecosystems, both terrestrial and aquatic, and we add up impacts of acidification and eutrophication.
- The numbers of hectares of ecosystem for which the critical loads are exceeded are evaluated for the whole of the EU15, non-EU and Europe. Hereby, regional differences in critical loads and the extent towhich the critical loads are exceeded are not accounted for.
- We use a single value for all ecosystems, irrespective of their characteristics and location. This simplification is characteristic for the valuation based on the implicit values of policy makers at the EU level.

- We assume that the costs as estimated by the technical-economic models are a good indicator for the WTP (e.g. Amann *et al*., 1999a and 1999b; Holland *et al*., 1999a and 1999b). Although it is an important element, policy makers also take other cost issues into account, including the impact of the measures on the economy, employment, and income distribution.
- We do not use marginal costs of single measures but the average costs of a marginal policy package. Although this value may be lower than the marginal costs of the individual measures, it better reflects the package deal in decision-making and its results are less sensitive to small changes in emission reduction scenarios or estimates of costs for single measures. Although it is true that marginal costs of additional measures are much higher, we cannot consider these higher costs to reflect the real WTP of politicians, as they were not willing to accept policy packages such as the initial NEC proposal (NEC+) with much higher marginal costs per tonne or hectare.
- The range of WTP values is determined by weight factors (0 or 1), representing the perception of policy makers of the importance of a certain effect during the negotiations on PRO and NEC.
- We have assumed that the ExternE-based[29] estimates for the effects of ozone are fully believed by policy makers and have been taken into account. The WTP is corrected for all of the impacts of ozone because the emission reduction programmes clearly define targets for AOT40 and AOT60.
- Setting targets for critical loads for acidification and eutrophication and for ozone (AOT40) also affects crop yields. Defining targets for SO_2 emission curbs is also beneficial for the protection of building materials. We have corrected the WTP for benefits for agriculture and building materials. This assumption is not that important as such benefits are relatively small.
- Although the studies indicate that there are big potential benefits from the emission reduction programmes on health impacts from secondary particles (aerosols), we have not used these data to correct the abatement costs for this benefit. A first reason is that it was not the objective of the agreements to tackle the issue of ambient particles. The major goal of the Gothenburg Protocol and the NEC directive for 2010 is, next to abatement of ground-level O_3, ecosystem protection, i.e. a 50% gap-closure of the accumulated exceedance of the critical loads for acidification and eutrophication. Although both emission reduction programmes mention the ‘additional’ benefit of a reduction of the formation of secondary particulate matter (aerosols) by SO_2 and NO_x emission curbs, this benefit very likely did not play a major role in the definition of the emission reduction targets for SO_2 and NO_x. This conclusion is based on the analysis of the official text of the Gothenburg Protocol and the legal text of the NEC directive. This assumption is also checked by the execution of a questionnaire with a small selection of key

[29] Although the data related to the benefits of the emission reduction scenarios are not identical to the ExternE data, they are based on similar methodologies, dose-response functions and valuation principles as the ExternE accounting framework. An important issue however is the presentation of the numbers in classes of uncertainty.

players that have been involved in the formulation of air pollution legislation. It is not possible to draw strict conclusions from this exercise but for the results obtained so far, we can conclude that secondary particles did play an important role during the negotiations on the Gothenburg Protocol and NEC directive but rather in an 'implicit, qualitative' way rather than in a 'tangible, quantitative' way. Second, although one may argue that the secondary particles effect had some impact on the negotiations, it is doubtful that these benefits got the same weighting as the ExternE numbers would suggest. The most important numbers (on chronic mortality) have a high uncertainty rating in cost-benefit analyses executed for the Gothenburg Protocol and the initial proposal on the NEC directive, as indicated in the reports of Holland *et al.* (1999a, 1999b). Third, if public health played a decisive role, and if the numbers were taken into account, policy makers should have decided on tighter emission standards.

- We have taken into account the benefits of ozone on public health and agriculture, but not the impacts of ozone on ecosystems.
- The 'reference' (REF) scenario has not been used to determine the range of the willingness-to-pay for improvement of ecosystems, as it includes measures focusing on other impact categories and the costs may not be comparable to these of other scenarios.
- We have based the WTP on the UN-ECE Gothenburg Protocol and the EU directive on NEC (NEC), as the policy makers have reached an agreement on these emission reduction programmes.
- The initial NEC proposal (NEC+) represents an upper margin for the WTP. The Council did not agree upon NEC+ but it was well founded and can be seen as a minimal interim goal if the EU long-term targets of no exceedance of the critical loads are to be reached in 2020.

10.3 Results and Discussion

10.3.1 WTP per hectare for improvement of ecosystems health.

In Table 10.1 the 'marginal' WTP range is presented and compared with the 'marginal' emission costs of the reference scenario (REF). It is important to note that this WTP range concerns the willingness-to-pay of the EU15 as a whole.

As there are arguments to base the WTP on each of the emission reduction programmes (PRO, NEC or NEC+), we report the range of possible values for WTP per hectare. The best estimate ranges from €63 to 350/hectare of ecosystems protected in Europe. If we calculate the WTP per hectare only for those ecosystems in the EU15, then these values go up from €338 to 674/ha. If we did not correct for the other benefits categories, then all these values would be higher (Figure 10.1).

Table 10.1 The 'marginal' WTP for the EU15 per hectare of ecosystem protected (in EUR/ha·year.

	REF (1)	*proxy WTP/ha* ↔ PRO (2)	NEC	*WTP/ha $_{MAX}$* NEC+ (3)
Corrected for other benefits*				
Per ha in Europe	466	**63**	/	**350**
Per ha in EU15	1.052	338	/	674
Uncorrected				
Per ha in Europe	469	145	/	463
Per ha in EU15	1.196	775	/	926

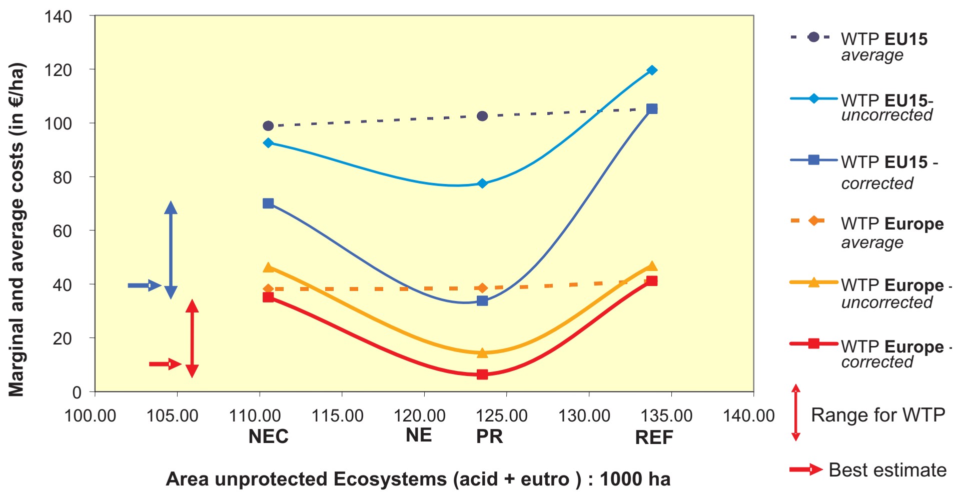
(*) Notes:

- Corrected for other benefits categories, according to the weight factors and based on abatement costs.
- The Table shows the full range. The numbers in bold indicate the range for the best estimate, following the assumptions as discussed in the text.
- Per ha in EU15: analysis based on impacts limited to area ecosystems in the EU15.
- Per ha in Europe: analysis based on impacts for area ecosystems all over Europe.

(1) Based on total costs for the reference scenario, compared to 1990.
(2) Based on total additional cost between PRO and REF.
(3) Based on total additional cost between NEC+ and PRO, calculated by extrapolation from benefits calculated at the European scale.

The lower range represents the additional costs to reach the Gothenburg Protocol, the higher values reflect the costs to meet the emissions reductions of the ambitious plan of the initial NEC proposal by the EC. As the policy makers have reached an agreement on the Gothenburg Protocol and NEC, we take these emission reduction programmes as the basis for our best estimate, for which we use a rounded number of €100/ha for the 'marginal' WTP per hectare of ecosystems protected in Europe. We assume that policy makers of the EU have the same WTP for improving ecosystems health all over Europe, including both EU15 and non-EU Europe. As the total number of hectares protected in the whole of Europe is more than twice the number of hectares protected in EU15, this leads to a lower estimate for WTP per hectare protected if the whole of the European area of ecosystems protected is evaluated. It does not, however, affect the shadow price expressed per tonne of pollutant. As the next step of the analysis, we multiply the WTP figure with the number of hectares affected by emissions from individual countries.

Analysing the costs curves from NEC to NEC+, the costs to reach the more ambitious emission reductions targets increase, especially if expressed as cost/tonne emission reductions (not presented in this report). This increase is less sharp if expressed per hectare protected, as the impact of one tonne of emission reduction on the number of hectares protected also increases.



Legend:

WTP EU15 / WTP Europe: *WTP per ha protected ecosystem within EU15 / Europe*

WTP-uncorrected: *WTP based solely on the abatement costs to be made by the EU15 member states per ha protected ecosystem*

WTP-corrected: *WTP per ha protected ecosystem based on the abatement costs to be made by the EU15 member states and corrected for other benefits (ground level ozone, benefits for agriculture and buildings) for the EU15.*

Average: *Average costs for emission reduction scenario compared to the base year 1990 (e.g. total costs of PRO compared to 1990 /total improvement of ha protected ecosystem of PRO compared to 1990).*

Figure 10.1 The 'marginal' and average cost of policy packages for protection of ecosystems (in €/extra hectare protected from acidification and eutrophication) for emission reduction goals as determined by the REF scenario, the Gothenburg Protocol (PRO), the NEC directive (NEC) and the proposal on the NEC directive (NEC+) for 2010.

One could argue that our best estimate is on the low end, because more costly measures have been decided upon in the reference scenario for many countries. It is difficult to interpret the costs of the REF scenario in comparison to the other scenarios. It is not clear whether this reflects a particularity of the models and data used, or whether it reflects the fact that the reduction measures put into practice are not based on the cost-optimal solutions (expressed in €/tonne emission reduction) but on a wider range of criteria, including, for example, effects on public health (transportation sector) or economic impacts of measures. Nevertheless, this may also illustrate that some member states have a higher WTP. Our approach cannot take this element into account.

On the other hand, one could also argue that the €100/hectare may be an upper estimate, as we did not correct for public health benefits of secondary particles, we did not take impacts of ozone on ecosystems into account, and the Protocol has not yet been ratified by most countries. For our purposes, the latter is not a real problem, as the NEC decision (at least) confirmed the targets for the EU member states.

The costs of the initial NEC proposal (NEC+) represents an upper limit for the WTP, as NEC+ was not agreed upon by the Council. Therefore, the upper limit for the 'marginal' WTP can be set to approximately €350/ha, taking into account all the hectares of ecosystems all over Europe.

10.3.2 Shadow prices for impacts on ecosystems from emissions of SO_2, NO_x and NH_3

We need to integrate our estimate on the marginal WTP in the impact pathway approach in order to calculate the shadow prices. Therefore, we first need to calculate the marginal impacts in physical terms, i.e. number of hectares of ecosystems for which the critical loads have been exceeded per additional tonne of SO_2, NO_x and NH_3 emitted.

Once the shadow prices are calculated, these data can be used to compare energy technologies and fuel cycles used in the EU. The figures are additive to the ExternE figures but are best separated, as they reflect another approach. Although detailed results based on the most recent critical loads data are not yet available, first evidence[30] suggests that, on average for EU 15, these impacts are unlikely to make a major contribution to the total damage cost, but may be significant for emissions from countries or regions with low impacts on human health and relatively high impacts on ecosystems. It has to be noted that the figures cannot be used in cost-benefit analysis or policy advice related to protection of ecosystems, as they are based on these policies.

10.3.3 Interpretation of policy decisions and referenda to derive a WTP

The evaluation has shown that, under certain assumptions, the costs of achieving the well-specified targets for acidification and eutrophication can be used to develop shadow prices for pollutants or specific impacts from pollutants. These shadow prices can be used to reflect these effects for comparison of technologies and fuel cycles.

The work undertaken shows that a simple analysis may not be correct, i.e. abatement costs for SO_2 and NO_x need to be corrected for other impacts. By analysing the decisions of policy makers in detail, shadow prices for exceedance of critical loads for eutrophication and acidification (ca. €100/hectare of exceeded area and year with a range of €60-350/ha year) have been developed.

[30] First estimates are based on critical load data from literature but these are outdated and do not match with the newer UN-ECE dataset used for the support of the Gothenburg Protocol and the NEC Directive (Hettelingh, private communication).

10.4 References

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11 Assessment of Uncertainty

11.1 Introduction

Since the uncertainties are large, an analysis of the uncertainties is crucial for the credibility of the results. To begin, one needs to identify and quantify the individual sources of uncertainty. It is appropriate to group them into different categories, even though there may be some overlap:

i. data uncertainty, e.g. slope of a dose-response function, cost of a day of restricted activity, and deposition velocity of a pollutant;
ii. model uncertainty, e.g. assumptions about causal links between a pollutant and a health impact, assumptions about form of a dose-response function (e.g. with or without threshold), and choice of models for atmospheric dispersion and chemistry;
iii. uncertainty about policy and ethical choices e.g. discount rate for intergenerational costs, and value of statistical life;
iv. uncertainty about the future, e.g. the potential for reducing crop losses by the development of more resistant species;
v. idiosyncrasies of the analyst, e.g. interpretation of ambiguous or incomplete information.

The first two categories (data and model uncertainties) are of a scientific nature. They are amenable to analysis by statistical methods, combining the component uncertainties over the steps of the impact pathway, in order to obtain formal confidence intervals around a mid estimate. For ethical choice and for uncertainty about the future a sensitivity analysis may be more appropriate, indicating how the results depend on these choices and on the scenarios for the future. If one is willing to assign probability distributions to the choices and scenarios, one can extend the statistical analysis to include the other categories as well. In this chapter we refrain from the subjective judgments that would be necessary to do that, and thus we address only the first two categories, noting that the full uncertainty could be significantly larger.

The uncertainties of environmental damages are far too large for the usual error analysis (using only the first term in a Taylor expansion) of physics and engineering. Rigorous systematic assessment of the uncertainties is difficult and few studies have attempted it. Most merely indicate an upper and a lower value but based on the range of just one input parameter or by simply combining the upper and lower bounds of each input, without taking into account the combination of uncertainties from several different inputs (e.g. atmospheric dispersion, dose-response function and monetary valuation). Many damage assessments involve so many different inputs that an analytical solution is not feasible and of the rare uncertainty analyses that have been done, almost all use Monte Carlo techniques and numerical calculations (see e.g.

Morgan *et al*., 1984; Morgan and Henrion, 1990). Such an approach is powerful, capable of treating any problem, albeit with heavy calculations. Above all, it is opaque: since it yields only numerical results, it is difficult to see how the numbers would change if some inputs are changed.

As an alternative we use a simpler and more transparent approach, based on lognormal distributions and multiplicative confidence intervals (Slob, 1994; Rabl and Spadaro, 1999). The justification lies in the observation that the calculation involves essentially a product of factors and that the resulting uncertainty of the product is approximately lognormal. 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). Compared to a Monte Carlo analysis, the lognormal approach yields simple typical answers that are easier to apply and communicate.

Some people have confused uncertainty and variability of the damage costs with site and local conditions. It is important to clearly distinguish between variability and uncertainty. Uncertainty is due to insufficient scientific knowledge and can be reduced by further research. Variability of impact with site is an intrinsic aspect of environmental impacts, independent of scientific uncertainties. Furthermore, the damage cost per kWh or per km varies with emissions. (In some cases variability can induce uncertainty, for example if the emissions per kWh vary between different power plants, the average damage of the power sector is uncertain if one does not have all the detailed emissions data for each power plant).

11.2 The Calculation of Damage Costs

11.2.1 The equations

If the dose-response function is stated in terms of ambient concentration as a concentration-response function (CRF), the impact rate due to exposure to an air pollutant can be written in the form

$$I(q) = \int dx \int dy \;\rho(\mathbf{x}) s_{CR}(\mathbf{x})\, c_{air}(\mathbf{x},q) \qquad (11.1)$$

where

$I(q)$ = impact rate [cases/yr], and
q = emission rate of pollutant [kg/s],
$c_{air}(\mathbf{x},q)$ = increase in concentration [$\mu g/m^3$] at a point $\mathbf{x}$ = (x,y) due to the emission q,
$\rho(\mathbf{x})$ = density of receptors (population, buildings, crops, …) [receptors/m^2] at $\mathbf{x}$,
$s_{CR}(\mathbf{x})$ = slope of CRF at $\mathbf{x}$ [(cases/yr)/(receptor·($\mu g/m^3$))].

Note that when inserting numbers one must be careful to convert all of them to a consistent set of units. For health impacts the CRFs of ExternE are linear and $c_{air}(\mathbf{x},q)$ should be the annual average concentration. For non-linear CRFs Eq.(11.1) should be used separately for different concentration ranges.

For numerical calculations the integral is replaced by a sum over grid cells. The impact per emitted quantity is the ratio of I and q, designated here by the symbol D (for damage, in physical units)

$$D = I(q)/q \,, \tag{11.2}$$

and multiplication by the unit cost p [€/case] yields the damage cost C in €/kg for the impact in question. Note that for primary pollutants the concentration increase $c_{air}(\mathbf{x},q)$ is proportional to the incremental emission q and therefore D and the damage cost are independent of q. For secondary pollutants, especially O_3 and nitrates, the relation can be non-linear if q is large; however, for marginal impacts q is small and one can still assume linearity.

The total damage cost of the pollutant is obtained by summing the individual C_i over all impacts i caused by this pollutant (for health the various impacts are called end-points).

$$C = \sum_i D_i \, p_i \,. \tag{11.3}$$

More than 95% of the total damage cost quantified by ExternE (for each pollutant with the exception of O_3 and greenhouse gases) is due to health impacts. Since the CRF slopes s_{CR} for health impacts are assumed to be independent of $\mathbf{x}$, s_{CR} can be taken outside the integral. Let us designate the remaining integral, divided by the emission rate, as exposure E per emitted quantity of pollutant

$$E = \int dx \int dy \;\; \rho(\mathbf{x})\, c_{air}(\mathbf{x},q)/q \,. \tag{11.4}$$

Thus for health impacts (and any other impact whose s_{CR} is independent of $\mathbf{x}$) the damage cost can be written as

$$C = E \sum_i s_{CRi} \, p_i \,. \tag{11.5}$$

Since E involves the integration of a complicated function, its uncertainty is difficult to evaluate. However, as shown in the following section, there is a very simple approximation, the "uniform world model" that yields results for typical situations.

With this model the uncertainty can be estimated with an explicit formula. A more detailed Monte Carlo calculation is described in section 11.3.5.

11.2.2 The "Uniform World Model"

Most policy applications concern pollution sources, the sites of which are not known in advance, and therefore one needs typical damage estimates rather than numbers for a specific installation. A simple and convenient tool for this purpose is the "uniform world model" (UWM), first presented by Curtiss and Rabl (1996) and further developed, with detailed validation studies, by Spadaro (1999), Spadaro and Rabl (1999) and Spadaro and Rabl (2002). More recently Spadaro and Rabl (2004) have extended it to toxic metals and their pathways through the food chain (the resulting equations are more complex than for inhalation and not shown here). The UWM is a product of a few factors; it is simple and transparent, showing at a glance the role of the most important parameters of the impact pathway analysis. It is exact for tall stacks in the limit where the distribution of either the sources or the receptors is uniform and the key atmospheric parameters do not vary with location.

It is convenient to express the concentration in terms of the flux $F_{dep}(\mathbf{x},q)$, defined as the rate [$\mu g/(m^2 \cdot s)$] at which the pollutant is removed from the atmosphere by dry and/or wet deposition

$$F_{dep}(\mathbf{x},q) = c_{air}(\mathbf{x},q)\, v_{dep}(\mathbf{x}) \tag{11.6}$$

where $v_{dep}(\mathbf{x})$ = total deposition velocity (dry plus wet) [m/s]. Using this definition to replace the concentration in Eq.(11.1) one obtains

$$I = \int dx\, dy\, s_{CR}(\mathbf{x})\, \rho(\mathbf{x})\, F_{dep}(\mathbf{x},q)/v_{dep}(\mathbf{x}) \tag{11.7}$$

which can be evaluated in closed form if $s_{CR}(\mathbf{x})$, $v_{dep}(\mathbf{x})$ and $\rho(\mathbf{x})$ are independent of $\mathbf{x}$, because the integral of the deposition flux is equal to the emission rate $\dot{m}$ by virtue of conservation of mass

$$q = \int dx \int dy\, F_{dep}(\mathbf{x},q)\ . \tag{11.8}$$

In a uniform world where $s_{CR}(\mathbf{x})$, $v_{dep}(\mathbf{x})$ and $\rho(\mathbf{x})$ have constant values s_{CR}, v_{dep} and ρ, the integral in Eq.(11.7) can therefore be evaluated in closed form, leading to the simple expression for the damage ("damage in uniform world")

$$D_{uni} = s_{CR}\, \rho / v_{dep} \tag{11.9}$$

This is called the "uniform world model" (UWM) for air dispersion (see Curtiss and Rabl, 1996; Spadaro, 1999).

In Figure 11.1 we compare Eq.(11.9) with the results of detailed site-specific calculations for about a hundred installations in many countries of Europe, as well as Southeast Asia and America (Spadaro, 1999; Spadaro and Rabl, 2002; additional calculations by Spadaro). Except for North America (where the detailed calculations were done with ExMod, a model similar to EcoSense), all of the detailed calculations were done with the EcoSense software of ExternE (Krewitt *et al*., 1995). UWM is so close to the average that it can be recommended for the calculation of typical values for emissions from tall stacks, more than about 50 m; for specific sites the agreement is usually within a factor of two to three. For ground level emissions in cities, the impact can be much larger than UWM because of the combination of high receptor densities with the high concentrations near ground level sources, but simple estimates can still be obtained by applying correction factors to UWM, discussed in the following section.

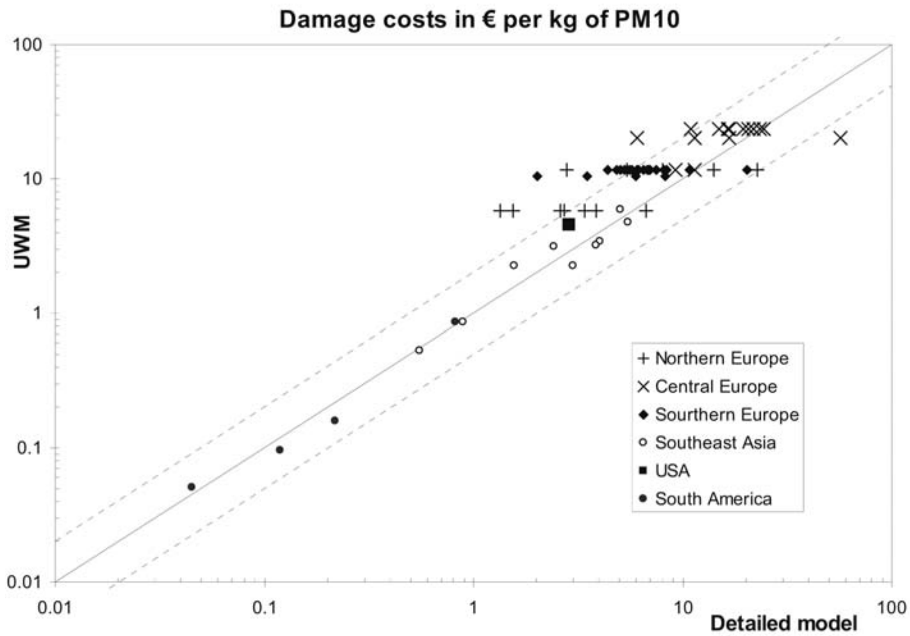

Figure 11.1 Comparison of UWM with detailed dispersion models. (The costs cannot be compared between America, Asia and Europe because of different assumptions about CRFs and unit costs).

The UWM involves the replacement of the average of a product by the product of the averages, an approximation that is justified to the extent that the factors are not correlated with each other and do not vary too much. In practice $F_{dep}(\mathbf{x}, \dot{m})$ varies strongly, being high near the source and decreasing with distance. For sources in large cities, this variation is correlated with the population density and so UWM underestimates the impact. For large cities far from the source the strong variation of $\rho(\mathbf{x})$ occurs in a region where the variation of $F_{dep}(\mathbf{x}, \dot{m})$ is slow, so their contribution is adequately taken into account in the average population density ρ and the acceptability of UWM is not affected.

The deposition velocity is derived by regression of dispersion data and the population density ρ is the average (land and water) within a radius of about 500 to 1000 km. The appropriate radius depends on the deposition velocity: a radius of 1000 km is appropriate for European conditions, but for regions with higher precipitation and hence more rapid deposition a smaller radius should be used. For example in Amazonia the deposition velocities are about three times larger than in Europe and a radius of 500 km is recommended. For central Europe the average population density is ρ = 80 pers/km^2 (land and water).

For the damage due to a secondary pollutant we note that Eq. (11.9) can be generalised to (Curtiss and Rabl, 1996)

$$D_{uni,sec} = \frac{s_{CR,\sec}\,\rho}{v_{dep,eff}} \tag{11.10}$$

where

$s_{CR,sec}$ is the CRF slope of the secondary pollutant, and

$v_{dep,eff}$ = an effective deposition velocity that accounts for the creation of the secondary pollutant and for the deposition.

The latter is given by

$$v_{dep,eff} = \frac{v_{1-2}}{v_{dep,1}\,v_{dep,2}} \tag{11.11}$$

with

$v_{dep,1}$ = deposition velocity of the primary pollutant,

$v_{dep,2}$ = deposition velocity of the secondary pollutant, and

$v_{1\text{-}2}$ = average of transformation velocity, defined as the regional average of

$$v_{1\text{-}2}(\mathbf{x}) = F_{1\text{-}2}(\mathbf{x})/c_1(\mathbf{x}) \tag{11.12}$$

where $F_{1\text{-}2}(\mathbf{x})$ = transformation flux and $c_1(\mathbf{x})$ = concentration of primary pollutant at $\mathbf{x}$.

In the region bounded by Sicily to the south, Portugal to the west, Scotland to the north and Poland to the east, the average population density is 80 persons/km^2. This is about half the average EU15 population density of 158 persons/km^2 per land area because it includes much water. Examining the results of detailed site-specific calculations for more than fifty installations in the EU15 countries, we have found that Eqs.(11.9) and (11.10), with the parameters in Table 11.1, do indeed provide representative results for power plants with typical stack heights.

Table 11.1 UWM parameters for several countries and regions, based on fits to EcoSense.

Source location	ρ [per km^2]	Primary pollutants $\mathbf{v_{dep}}$ of Eq.(11.9) [cm/s]			Secondary pollutants $\mathbf{v_{dep,eff}}$ of Eq.(11.10) [cm/s]	
		PM_{10}	SO_2	NO_2	$NO_2 \rightarrow$ Nitrates	$SO_2 \rightarrow$ Sulphates
EU (central)	80	0.67	0.73	1.47	0.71	1.73
Scandinavia	20	0.67	0.73	1.47	0.71	1.73
Cyprus	54	0.64	0.77	1.27	0.84	1.36
Czech Rep.	111	0.89	0.87	1.04	1.26	2.15
Estonia	43	0.93	1.00	1.67	1.29	1.35
Hungary	101	0.85	0.94	1.53	1.01	1.77
Latvia	55	0.93	1.00	1.67	1.29	1.35
Lithuania	62	0.93	1.00	1.67	1.29	1.35
Poland	100	0.86	0.89	1.05	1.29	1.98
Portugal	89	0.86	0.90	0.96	1.23	2.00
Slovakia	101	0.85	0.94	1.53	1.01	1.77
Slovenia	105	0.85	0.94	1.53	1.01	1.77
Spain	55	0.67	0.73	1.47	0.71	1.73
China	200	0.74	0.66	0.96	0.97	0.90

The UWM is the simplest of several models for approximate assessments contained in the RiskPoll software package (developed by J. Spadaro and downloadable from www.arirabl.com).

11.2.3 Variation with stack height and local conditions

Obviously UWM is not sufficiently detailed to account for aspects such as stack height or proximity of large population centres. In this regard there is a large difference between primary and secondary pollutants. The impact of primary pollutants varies strongly with stack height and local receptor distribution because the ground level concentration in the vicinity of the source is very sensitive to these aspects. In contrast, for secondary pollutants the impact is very little affected by local conditions because the creation of secondary pollutants is relatively slow: for secondary particulate matter it takes place over tens to hundreds of km; the formation of O_3 is somewhat faster, over distances of km to tens of km. In particular the variation of impact with stack height is completely negligible for secondary particulate matter.

For an indication of the variation of the impact with local conditions for primary pollutants, Figure 11.2 shows the variation of the impact of SO_2 (as primary pollutant) with stack height for five sites in France. Of course, the impact is largest for a source near a very large city, Paris (population of Greater Paris is about 10 million). If the source were in Paris itself rather than some 40 km away as for Porcheville in Figure 11.2, the variation with height would be even more pronounced, being as large as 15 times the UWM value.

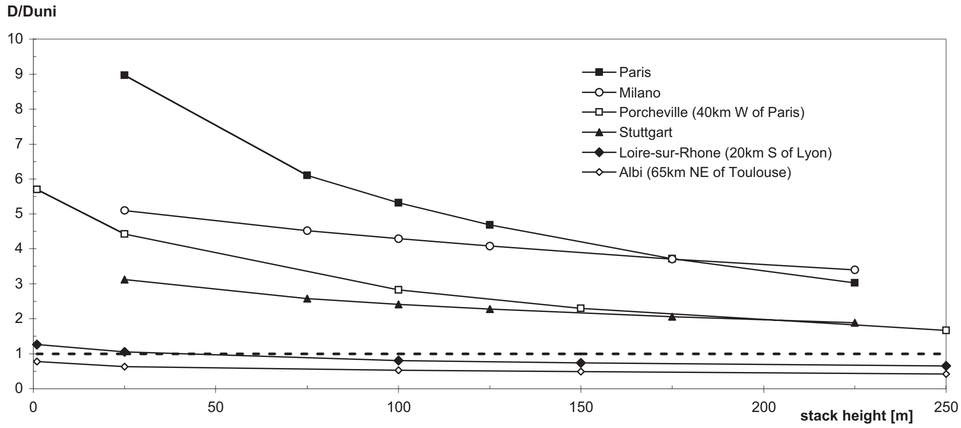

Figure 11.2 Dependence of damage D on site and on height of source for a primary pollutant with linear CRF, for six sites, in units of D_{uni} for uniform world model (UWM) Eq.(11.9) with ρ = 80 persons/km^2. Except for ground level sources, the plume rise for typical power plant conditions is taken into account.

As for the variation with stack conditions (stack height h, temperature T, exhaust velocity v) and local conditions (proximity of big city, local climatic conditions), the typical values obtained by UWM or the multisource version of EcoSense need to be multiplied by the correction factors shown in Table 11.2. For example, the cost/kg of $PM_{2.5}$ emitted by a car in Paris is about $6 \cdot 15 = 90$ times D_{uni} (a factor 6 for variation with site and a factor 15 for variation with stack height).

Table 11.2 Correction factors for estimating site-specific damage costs on the basis of typical values calculated with UWM.

Pollutant	Correction factors
Globally dispersing pollutants such as CO_2	1 for all sites
Pollutants for which non-inhalation pathways dominate (dioxin, Pb, As)	≈ 0.7 to 1.5
Nitrates (due to NO_x) and sulphates (due to SO_2)	≈ 0.5 to 2.0 for site (higher values if large population near source)
Inhalation impacts of primary pollutants	≈0.5 to 6 for site (higher values if large city near source), ≈0.6 to 15 for stack height (higher values for low stacks, especially in big city).

How can the damage cost estimates be transferred from one site to another? To the extent that more than 95% of the costs arise from health impacts, they are proportional to the size of the affected population weighted by the respective concentration increments. For precise results one would have to repeat the analysis based on local meteorological and population data, but for a rough first estimation one can use the following rules of thumb (Spadaro, 1999; Spadaro and Rabl, 1999):

- for primary pollutants emitted by vehicles in cities, the damage cost is roughly proportional to the population of the conurbation;
- for secondary pollutants the damage cost is roughly proportional to the average regional population density within a radius of 500 to 1000 km; the radius of the region is smaller in regions with high precipitation such as Brazil.

In regions where the unit costs are different, these would have to be adjusted as well.

11.3 Uncertainty of Sums and Products

11.3.1 Sums

The UWM for the damage cost of a single impact involves a simple product (and for most pollutants a single impact, mortality, contributes more than two thirds of the total damage cost). If several impacts make a significant contribution, one also has to sum over such products for the total damage cost. For sums and products an analytical solution is possible.

To begin, consider the sum

$$y = x_1 + x_2 + ... + x_n \tag{11.13}$$

of uncorrelated random variables x_i. The mean of y is

$$\mu_y = \mu_{x1} + \mu_{x2} + \ldots + \mu_{xn} \tag{11.14}$$

where the μ_{xi} are the means of the x_i. The standard deviation σ_y of y is given by the usual quadratic combination

$$\sigma_y^2 = \sigma_{x1}^2 + \sigma_{x2}^2 + ... + \sigma_{xn}^2 \tag{11.15}$$

of the standard deviations σ_{xi} of the x_i. Even though these relations are exact, regardless of the size of the standard deviations, they do not yield an interpretation of σ_y in terms of confidence intervals. For that one also needs the probability distribution of y. Fortunately in many cases of practical interest the distributions are approximately Gaussian (also called normal). In particular, in the limit where the number of terms in the sum becomes large, the central limit theorem of statistics implies that the distribution of y approaches a Gaussian regardless of the individual distributions of the terms in the sum. In practice the distribution of y is close to a Gaussian unless one or several of the terms have distributions that have large standard deviations and are very different from Gaussian. When the distribution of y is nearly Gaussian, one can say that $[\mu_y - \sigma_y, \mu_y + \sigma_y]$ is approximately the 68% confidence interval and $[\mu_y - 2\sigma_y, \mu_y + 2\sigma_y]$ approximately the 95% confidence interval.

11.3.2 Products

These considerations apply also to the product z of uncorrelated variables x_i

$$z = x_1\, x_2\, x_3 \ldots x_n \tag{11.16}$$

if one looks at the logarithm. The mean of the logarithm of a random variable is the logarithm of the geometric mean μ_g; specifically, if p(z) is the probability distribution of z, the geometric mean is given by

$$\ln(\mu_{gz}) = \int_0^\infty p(z) \ln(z)\, dz \tag{11.17}$$

Since the mean of $\ln(\mu_{gz})$ is the sum of the logarithms of the geometric means μ_{gxi} of the x_i, μ_{gz} is given by the product

$$\mu_{gz} = \mu_{gx1}\, \mu_{gx21} \ldots\ \mu_{gxn}\,. \tag{11.18}$$

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

$$[\ln(\sigma_{gz})]^2 = \int_0^\infty p(z)\,[\ln(z) - \ln(\mu_{gz})]^2\,dz \tag{11.19}$$

and analogously for the x_i. With independence of the distributions, one finds that the geometric standard deviation σ_{gz} of the product z is given by

$$[\ln(\sigma_{gz})]^2 = [\ln(\sigma_{gx1})]^2 + [\ln(\sigma_{gx2})]^2 + ... + [\ln(\sigma_{gxn})]^2\ . \tag{11.20}$$

11.3.3 The lognormal distribution

The lognormal distribution of a variable z is obtained by assuming that the logarithm of z has a normal distribution (see e.g. Morgan and Henrion, 1990). Invoking the central limit theorem for the product z, one sees that the lognormal distribution is the "natural" distribution for multiplicative processes, in 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 for a few factors, provided the distributions with the largest spread are not too far from lognormal. For many environmental impacts the lognormal model for the result is quite relevant because the impact is a product of factors and the distributions of the individual factors are not too far from lognormality. All one has to do is to estimate the geometric standard deviations of the individual factors and combine them according to Eq.(11.20).

It is instructive to derive the lognormal probability density distribution. Let u be a Gaussian variable with mean ξ and standard deviation ϕ. Its probability density distribution is given by

$$g(u) = \frac{1}{\phi\sqrt{2\pi}} \exp\left[-\frac{(u-\xi)^2}{2\phi^2}\right]. \tag{11.21}$$

normalised to unity when integrated over u from $-\infty$ to $+\infty$. With a change of variable

$$u = \ln z \tag{11.22}$$

the normalisation integral becomes

$$\int_0^\infty \frac{g(\ln(z))}{z}\,dz = 1 \tag{11.23}$$

which allows one to interpret the function

$$p(z) = \frac{g(\ln(z))}{z} = \frac{1}{\phi z \sqrt{2\pi}} \exp\left[-\frac{(\ln(z)-\xi)^2}{2\phi^2}\right] \tag{11.24}$$

as the probability density of a new distribution between 0 and $+\infty$. This is the lognormal distribution. An example is plotted in Figure 11.3. If a distribution is lognormal, the geometric standard deviation indicates multiplicative confidence intervals, analogous to the additive confidence intervals of the Gaussian distribution. One can show that, for the lognormal distribution, the geometric mean μ_g is equal to the median. If a quantity with a lognormal distribution has been found to have a geometric mean μ_g and a geometric standard deviation σ_g, the probability is approximately 68% for the true value to be in the interval $[\mu_g/\sigma_g, \mu_g\,\sigma_g,]$ and 95% for it to be in the interval $[\mu_g/\sigma_g^2, \mu_g\,\sigma_g^2]$.

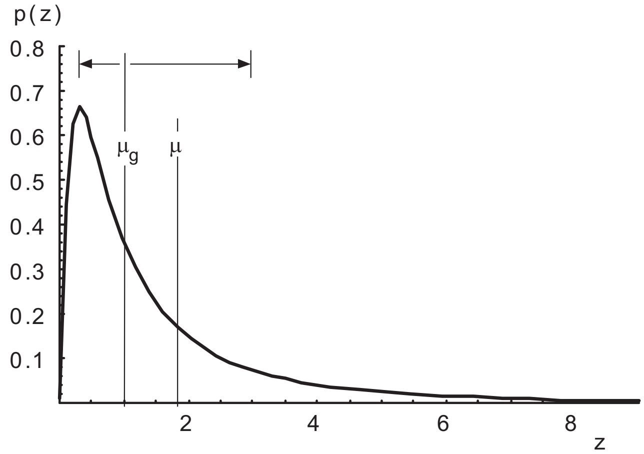

Figure 11.3 Probability density of lognormal distribution with $\mu_g = 1$ and $\sigma_g = 3$. Mean $\mu = 1.83$. The arrows indicate the 68% confidence interval (1 σ_g interval).

One can show that the ordinary mean μ and standard deviation σ of the lognormal variable z are given by

$$\mu = \langle z \rangle = \exp(\xi + \phi^2/2) \tag{11.25}$$

and

$$\sigma^2 = \langle (z - \langle z \rangle)^2 \rangle = [\exp(\phi^2) - 1]\exp(2\,\xi + \phi^2) \tag{11.26}$$

(the notation <…> indicates expectation value). The geometric mean μ_g and the geometric standard deviation σ_g are related to the parameters ξ and ϕ by

$$\mu_g = \exp(\xi) \quad \text{and} \quad \sigma_g = \exp(\phi) \,. \tag{11.27}$$

Given any one of the pairs $\{\xi, \phi\}$, $\{\mu, \sigma\}$ or $\{\mu_g, \sigma_g\}$, the others can be determined by means of Eqs.(11.25) to (11.27). In particular it is easy to calculate μ and σ from μ_g and σ_g

$$\mu = \mu_g \ \exp\left(\frac{[\ln(\sigma_g)]^2}{2}\right) \tag{11.28}$$

and

$$\sigma = \mu \ \sqrt{(\mu/\mu_g)^2 - 1} \,. \tag{11.29}$$

11.3.4 Sums of products

According to Eq.(11.5) the damage costs of ExternE involve sums of products. For that the analysis becomes more complicated. It is easy to combine the geometric standard deviations of s_{CRi} and p_i to get that of the product $s_{CRi}\ p_i$, but for their sum one needs the ordinary standard deviations of each product $s_{CRi}\ p_i$. They can be obtained by means of Eqs.(11.25) to (11.27). Finally, having determined the ordinary standard deviation of the sum, one can use these equations again to find the corresponding geometric standard deviation. However, this latter step is not rigorous because Eqs.(11.25) to (11.27) are exact only for a lognormal distribution. Thus the analytical solution is more complicated and only approximate, and with so many parameters (for each of the end-points) that the result is no longer very transparent.

Fortunately, in practice the calculations can often be greatly simplified by noting that, thanks to the quadratic combination of error terms, small terms in the quadratic sum can be neglected with negligible effect on the overall result. It is instructive to illustrate this phenomenon with the examples in Table 11.3. The magnitude of x_1 and x_2 in part b) is chosen to correspond roughly to the relative contributions of mortality and the other impacts to the total damage cost of PM, NO_x and SO_2. If both terms have the same relative error $\sigma_i/x_i = 50\%$, the contribution of the error of the smaller term to the total error of the sum is quite small. Even in part b) the difference between the relative errors of the larger term (50%) and of the sum (37%) does not appear significant in view of the subjectivity of any uncertainty estimate in this domain. Therefore we consider only the uncertainty of mortality and take its σ_g as an appropriate estimate of the uncertainty of the total damage cost of these pollutants.

Table 11.3 Examples of combination of errors in a sum of two terms (x_1 and x_2), each with a relative error σ_i/x_i = 50%.

a) First term is 80% of total.

	Sum	σ	σ^2	Relative error
x_1	0.8	0.4	0.16	50%
x_2	0.2	0.1	0.01	50%
Total	1.0	$\sqrt{0.17} = 0.41$	0.17	41%

b) First term is 65% of total.

	Sum	σ	σ^2	Relative error
x_1	0.65	0.325	0.11	50%
x_2	0.35	0.175	0.03	50%
Total	1.0	$\sqrt{0.14} = 0.37$	0.14	37%

11.3.5 A Monte Carlo analysis of exposure

During the recent ExternE-Pol (Rabl *et al*., 2004) phase of ExternE, we have estimated the uncertainty of exposure by means of a Monte Carlo calculation, taking into account the uncertainties of the numerous input data. Probability distributions are used for the possible values of the input parameters. Since some of the distributions are not well known, several possible cases are considered. Only dispersion is taken into account, without chemical reactions.

The analysis starts from the mass balance for the average pollutant concentration in a column of air that moves with the wind from source to receptors. For an initial analysis several assumptions are made to calculate the ground level concentrations; they are generally made by models that calculate collective exposure and are believed to be sufficiently realistic for that purpose:

- A1: the pollutant moves along straight trajectories away from the source;
- A2: the wind speed does not vary with height;
- A3: wind speed and atmospheric stability class are constant for a puff moving along its trajectory;
- A4: there is no exchange with the upper atmosphere above the mixing layer height;
- A5: the distributions of the parameters are statistically independent;

- A6: for the ratio of the ground level and the column-average concentrations, one can take the ratio of a Gaussian plume dispersion model, multiplied by a random number with a lognormal distribution.

Then a sensitivity analysis is carried out to test what happens when these assumptions are relaxed. This approach provides a model-independent assessment of the uncertainty of any dispersion model that satisfies the assumptions, including EcoSense.

Results have been obtained for power plants (with stack height 75 m and typical plume rise) at three locations: an extremely large population centre (Paris), an intermediate site (Lauffen near Stuttgart) and a rural site (Albi in the Southwest of France). The uncertainty of the collective exposure, expressed as geometric standard deviation σ_g, ranges from about 1.2 for Paris and 1.5 for Stuttgart to about 1.9 for Albi. The uncertainty is larger for rural sites because for a rural site the regional impacts dominate and the regional impacts are very sensitive to the assumptions about the deposition velocity, whereas deposition is almost negligible in the local zone (for PM_{10}, SO_2 and NO_x). Since most pollution sources tend to be located more in or around cities than in rural areas, we assume a σ_g of 1.5 for dispersion, significantly lower than the value of 2 that we had taken in an earlier analysis Rabl and Spadaro (1999).

11.4 Component Uncertainties and Results

11.4.1 Atmospheric models

For the uncertainty of atmospheric models a geometric standard deviation in the range from two to five is sometimes cited, but without making a distinction between episodic values and averages over space or time. In fact, atmospheric models are far more accurate for averages than for episodic values. This is an important consideration since for policy applications, which are of interest here, one needs long-term average values rather than episodic values. For example, the European tracer experiment (ETEX) (Van Dop *et al*., 1998) has provided validation for a variety of dispersion models, but on an episodic basis; the relatively large discrepancies between measured and calculated values are therefore no indication of the accuracy that can be expected for long-term averages.

Among the many parameters and input data of an atmospheric model, most have only a relatively minor effect on the calculation of long-term average concentrations. To see which parameters are the most important, it is instructive to look at the "uniform world model" described in section 11.2.2, because it yields the damage costs for typical conditions. In the "uniform world model" the key parameters of a dispersion model are those that affect the depletion velocity, defined by Eqs.(11.6) and (11.11). For an indication of the kind of distribution that can be expected, we show in Figure 11.4 a histogram of dry deposition velocity data for 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 is due to different surface materials, atmospheric conditions and variation with time of day and year. More recent data may have a smaller standard deviation but we have not been able to find a survey as comprehensive as that of Sehmel.

For particles we refer to Fig. 19.3 of Seinfeld and Pandis (1998), which likewise suggests a lognormal distribution. Dry deposition of small particles has been reviewed by Nicholson (1988), who points out the large variability of measured deposition velocities with the nature of the surface and the conditions of the observation. The spread of values seems to be comparable to Figure 11.4 for SO_2.

Deposition velocities for reactive nitrogen compounds have been reviewed by Hanson and Lindberg (1991); here the variability with the conditions of the absorbing surface is further enhanced by the high chemical reactivity of nitrogen compounds.

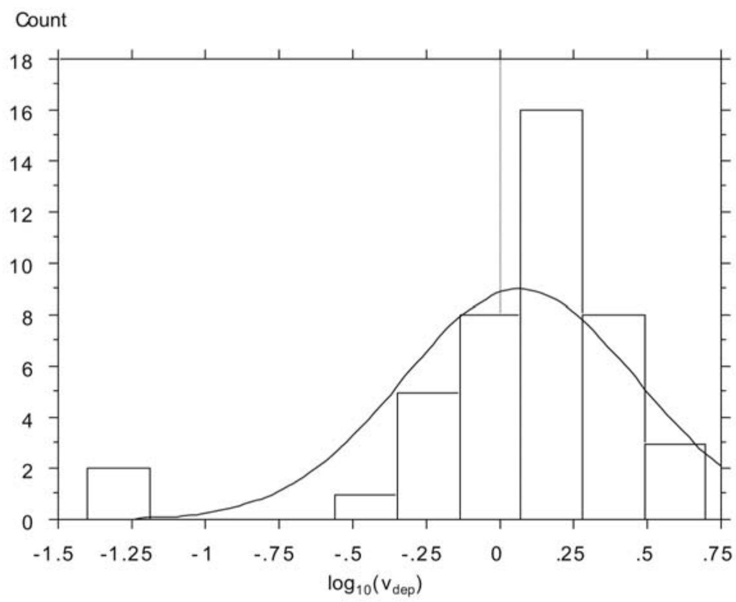

Figure 11.4 Distribution and lognormal fit of maximum values, in the review of Sehmel (1980), for dry deposition velocity [in cm/s] of SO_2 over different surfaces.

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 the ECOSENSE model (Krewitt *et al*., 1995). Thus the uncertainty of total deposition in Europe appears to be significantly smaller than suggested by dry deposition data.

Based on the Monte Carlo calculation described in section 11.3.5, we are led to suggest a σ_g around 1.5 for dispersion modelling of primary non-reactive air pollutants. For the dispersion of NO_x and SO_2 we take a somewhat larger σ_g because their impacts occur mostly at greater distances from the source, thus rendering their dispersion aspects more like the rural situation for PM. These numbers are consistent with estimates by McKone and Ryan (1989). For secondary pollutants there is additional uncertainty due to chemistry, especially in the case of ozone, but the uncertainty due to inaccuracies in the spatial distribution of concentration values relative to receptors is much smaller because secondary pollutants form only gradually at distances removed from the source. Since the chemical reactions depend on the background concentrations which are not sufficiently well known, we also introduce a σ_g for the effect of background emissions.

To sum up this section, we assume

- $\sigma_g = 1.5$ for the dispersion of non-reactive primary pollutants
- $\sigma_g = 1.7$ for the dispersion of SO_2 and sulphates
- $\sigma_g = 1.2$ for the formation of sulphates from SO_2
- $\sigma_g = 1.05$ for the effect of background emissions on the formation of sulphates from SO_2
- $\sigma_g = 1.7$ for the dispersion of NO_x and nitrates
- $\sigma_g = 1.4$ for the formation of nitrates from NO_x
- $\sigma_g = 1.15$ for the effect of background emissions on the formation of nitrates from NO_x

11.4.2 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 CRFs for certain health impacts from the classical 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.

The confidence intervals of CRFs for health impacts 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.

If one knows the probability distribution of the residues in the respective studies, one could calculate the geometric standard deviation exactly from its definition in Eq.(11.19). If one does not, but the reported confidence intervals are symmetric, it is reasonable to assume a gaussian distribution. Strictly speaking, the resulting σ_g is complex because the gaussian is non-zero at negative values. However, negative values are not plausible on physical grounds (for health impacts of air pollutants a beneficial effect is not plausible) and the distribution should be cut off at zero. Furthermore, if one uses only CRFs that are statistically significant at the 95% level, the contribution of the negative values represents at most 2.5% of the normalisation integral of the Gaussian, and the effect on the resulting σ_g would be negligible.

A much simpler alternative is the following approximation. Suppose that $\pm\delta/2$ corresponds to a 68% confidence interval, as for a gaussian distribution. Then one fits a corresponding lognormal distribution such that its 68% confidence interval equals [μ-$\delta/2$, $\mu+\delta/2$], which yields σ_g as

$$\sigma_g = \sqrt{\frac{\mu+\delta/2}{\mu-\delta/2}}. \tag{11.30}$$

We have compared the more rigorous approach with this approximation for the example of the mortality risk due to $PM_{2.5}$ reported by Pope *et al*. (1995). For the quantity (relative risk – 1) at the highest concentration, we have $\mu = 0.17$ and $\delta = 0.085$. Inserting these numbers into Eq.(11.30) one finds $\sigma_g = 1.29$, whereas the more rigorous approach, via Eq.(11.30), yields $\sigma_g = 1.32$. The difference is not significant.

We have evaluated Eq.(11.30) for all the CRFs of ExternE (European Commission, 1995 and 1999) for NO_x, SO_2, PM and O_3; typically σ_g is in the range 1.2 to 1.8.

For chronic mortality one also needs to determine the relation between the YOLL (years of life lost) and the change in the age-specific mortality rate that has been reported by studies of chronic mortality. Leksell and Rabl (2001) have examined the uncertainties of this calculation; their results suggest a σ_g of 1.3 for the calculation of the YOLL, given the relative risk.

There is, however, another type of uncertainty due to the difference between the PM in ambient air on which epidemiology is based and the primary and secondary PM in the damage calculations, as explained in the chapter on health impacts. Ambient PM is a mix of primary PM from combustion and secondary PM, especially nitrates and sulphates. For the damage calculations one needs assumptions about the relative toxicity of the different components of ambient PM. The uncertainty of these assumptions is difficult to estimate. For the calculations of σ_g we assume that the corresponding uncertainty might be a σ_g of 1.5 for primary particles and 2 for nitrates and sulphates.

To sum up this section, we assume

σ_g = 1.2 to 1.8 for the morbidity CRFs due to primary particles
σ_g = 2 for the morbidity CRFs due to nitrates and sulphates
σ_g = 1.5 for the relative mortality risk due to primary particles
σ_g = 2 for the relative mortality risk due to nitrates and sulphates
σ_g = 1.3 for the calculation of the YOLL for a given mortality risk.

11.4.3 Monetary valuation

Some physical impacts can be easily valued by their price on the market, e.g. the price of crops. There is little uncertainty in these prices as quoted at a 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.

Nonmarket goods are difficult to value economically. This is especially true for the reference value for the protection of human lives, often called value of a prevented fatality (VPF). It involves an ethical choice and there is now 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 VPF results from various studies of individual preferences tends to be lognormal, as illustrated for example by Figure 11.5 which is based on the Ives, Kemp and Thieme (1993) survey of 78 VPF studies published between 1973 and 1990. Figure 11.5 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.

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 11.4 which summarises the distributional characteristics of Ives, Kemp and Thieme (1993). The spread is so large, with a geometric standard deviation of 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 11.4, it is fairly close to the geometric mean.

There appears to be an emerging political consensus in Europe and North America that a value in the range of 1 to 5 MECU is reasonable. Thus we will assume a σ_g of 2 for the VPF. We also take σ_g of 2 for the value of a life year (VOLY) in view of the results of the VOLY study carried out in the NewExt phase of ExternE (ExternE, 2004). Since the unit cost of chronic bronchitis is based on contingent valuation, just like VOLY, we assume the same σ_g.

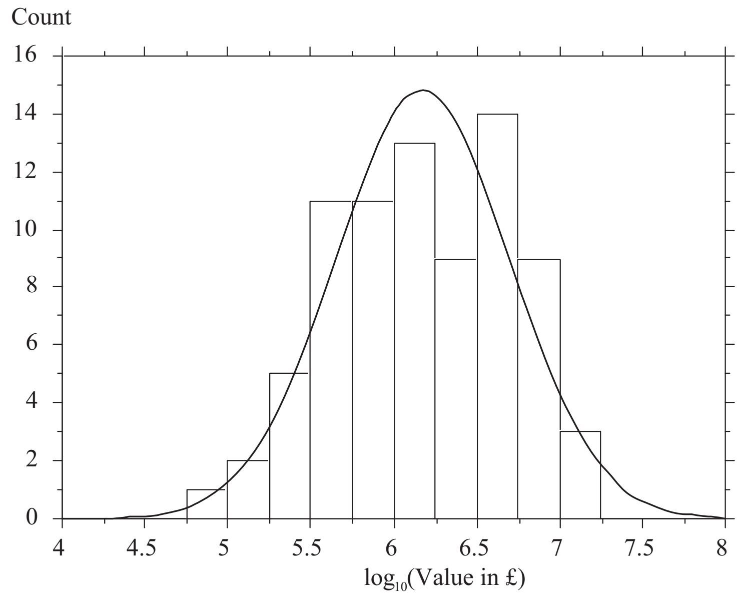

Figure 11.5 Example of lognormal distribution for economic valuation: reference value for protection of human life, in $£_{1990}$, as determined by 78 studies reviewed by Ives, Kemp and Thieme (1993), histogram and lognormal fit plotted on log scale.

Table 11.4 Uncertainty of the value of statistical life: distributional characteristics from the survey by Ives, Kemp and Thieme (1993) – $£_{1990}$ 1 = $ 1.78.

Mean	£ 2.76 million
Standard deviation	£ 3.00 million
Median	£ 1.59 million
Geometric mean μ_g	£ **1.49** million
Geometric standard deviation σ_g	**3.4**

The uncertainties for other costs, such as hospitalisation and agricultural losses, are much smaller because such data can be determined from the market; we estimate σ_g of 1.1 to 1.3.

To sum up this section, we assume

σ_g = 1.1 to 1.3 for market prices

σ_g = 2 for chronic bronchitis and for mortality (VPF and VOLY).

11.4.4 Total σ_g of damage costs

Table 11.5 shows the assumptions for the component uncertainties and the result for the σ_g of the damage cost for mortality. For sulphates and nitrates ExternE assumes the same CRFs as for PM (apart from an overall scale factor), therefore the contributions to the uncertainty are the same for each of these pollutants, with the exception of

i) atmospheric dispersion and chemistry (we assume different geometric standard deviations for PM, NO_x and SO_2);
ii) the toxicities of primary PM, sulphates and nitrates relative to ambient PM10, as discussed in the chapter on health impacts.

Table 11.5 Uncertainty of damage cost estimates per kg of pollutant for mortality. Sample calculations of geometric standard deviation σ_g, inserting the component uncertainties σ_{gi} into Eq.(11.20) ($(\ln(\sigma_g))^2$ = sum of the $\ln(\sigma_{gi})^2$) The relative contributions of the σ_{gi} to total are indicated under $\ln(\sigma_{gi})^2$.

1		*lognormal?*	**σ_{gi} PM**	*$\ln(\sigma_{gi})^2$*	**σ_{gi} SO_2** via sulphates	*$\ln(\sigma_{gi})^2$*	**σ_{gi} NO_x** via nitrates	*$\ln(\sigma_{gi})^2$*
2	*Exposure calculation*							
3	Dispersion	*yes*	1.5	*0.164*	1.5	*0.164*	1.5	*0.164*
4	Chemical transformation	*yes*	1	*0.000*	1.2	*0.033*	1.4	*0.113*
5	Background emissions	*no*	1	*0.000*	1.05	*0.002*	1.15	*0.020*
6	*CRF*							
7	Relative risk	*no*	1.3	*0.069*	1.3	*0.069*	1.3	*0.069*
8	Toxicity of PM components	*?*	1.5	*0.164*	2	*0.480*	2	*0.480*
9	YOLL, given relative risk	*no?*	1.3	*0.069*	1.3	*0.069*	1.3	*0.069*
10	*Monetary valuation*							
11	Value of YOLL (VOLY)	*yes*	2	*0.480*	2	*0.480*	2	*0.480*
12	**Total**		**2.65**	***0.95***	**3.13**	***1.30***	**3.26**	***1.40***

The resulting geometric standard deviations are 2.65 for primary PM, 3.13 for SO_2 and 3.26 for NO_x. We show three significant figures only to bring out the differences between these pollutants and the larger uncertainties of the secondary pollutants. But in view of the subjective and rather uncertain assumptions we had to make about the component uncertainties, we believe that it is best to simply sum up the results by saying that the geometric standard deviation of these damage costs is approximately 3. For pollutants such as dioxins, As and Pb whose impacts come mostly from ingestion, we estimate, very roughly, that the geometric standard deviation is around 6.

11.5 Placement of the Confidence Intervals

A comment is required about the placement of the confidence intervals relative to the damage cost estimates. In fact, one needs to consider whether the key parameters of the calculations have been estimated as means, medians or something else, for instance modes (= point where the probability distribution of possible parameter values has its maximum). An inquiry into the practice of researchers in this field leads to the conclusion that the typical choice is the mean. For example, the value of a life year is based on a mean of the contingent valuation results in different EU countries (ExternE, 2004).

Since the confidence intervals that have been estimated for the damage costs are symmetric around the median (=geometric mean for lognormal distribution) on a logarithmic scale, their placement relative to the quoted damage costs has to be modified. For a lognormal distribution the ratio of mean μ and median μ_g is given by

$$\mu/\mu_g = \exp(0.5\ \phi^2) \quad \text{with } \phi = \ln(\sigma_g)\ . \tag{11.31}$$

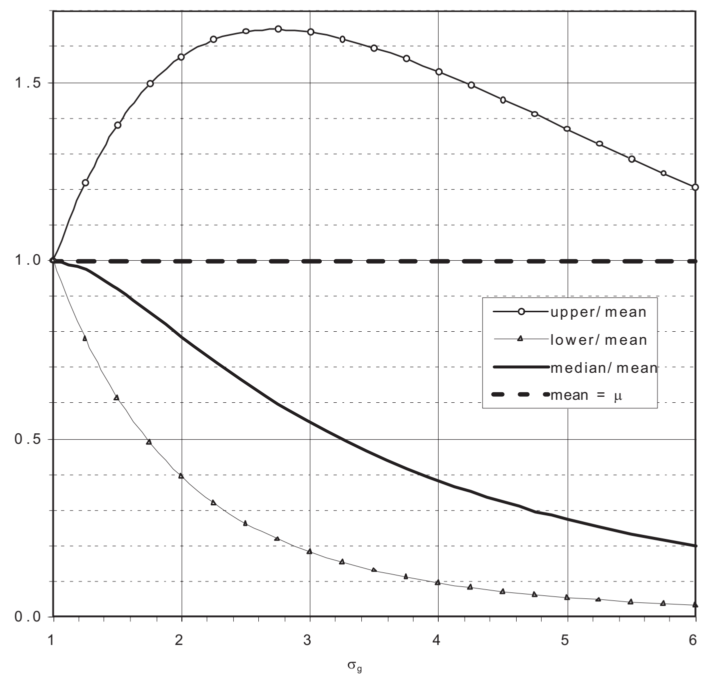

Figure 11.6 Median (μ_g), upper bound ($\mu_g \times \sigma_g$) and lower bound (μ_g/σ_g) relative to mean μ, as function of the geometric standard deviation σ_g for a lognormal distribution. mean μ = 1.

There is a sizeable difference between median and mean, as can be seen in Figure 11.6 where the ratios median/mean, upper/mean and lower/mean are plotted as function of the geometric standard deviation σ_g for a lognormal distribution.

To see what this implies for the placement of the upper and lower bounds, let us consider the numbers for PM damage cost according to the UWM. If each factor in the formula had a ratio mean/median according to its geometric standard deviation σ_g (as per Figure 11.6), the mean/median ratio of the damage cost would be given directly by Eq.(11.31). For example, if $\sigma_g = 3$, ϕ is 1.099 and the mean/median ratio $\mu/\mu_g = 1.83$.

Table 11.6 Placement of the 68% confidence intervals.

1		**PM**	**SO_2** via sulphates	**NO_x** via nitrates	**Explanation**	**Typical** inhalation	**Typical** ingestion
2	σ_g exposure (v_{dep})	1.50	1.56	1.72	Eq.(11.20) with σ_{gi} of Table 11.5 lines 3-5		
3	ϕ exposure	0.405	0.447	0.545	$\phi = \ln(\sigma_g)$		
4	mean/median, exposure	1.09	1.11	1.16	Eq. (11.31)	1.1	1.1
5	**σ_g total**	**2.65**	**3.13**	**3.26**	Table 11.5 line12	**3**	**6**
6	ϕ total	0.973	1.140	1.181	$\phi = \ln(\sigma_g)$	1.10	1.79
7	mean/median, total uncorrected	1.61	1.91	2.01	Eq.(11.31)	1.83	4.98
8	mean/median, total corrected	1.36	1.57	1.49	line 7/(line 4)^2	1.51	4.12
9	median/mean	0.73	0.64	0.67	1/line 8	0.66	0.24
10	**lower/mean**	**0.28**	**0.20**	**0.21**	line 9/line 5	**0.22**	**0.04**
11	**upper/mean**	**1.94**	**1.99**	**2.18**	line 9 · line 5	**2.00**	**1.44**

However, the depletion velocity v_{dep} is in the denominator rather than the numerator of the UWM of Eqs.(11.9) and (11.10). The mean/median ratio of $1/v_{dep}$ is in fact the inverse of the mean/median ratio of v_{dep}. We take σ_g of the depletion velocity v_{dep} as the total for atmospheric modelling, namely the combination of σ_g for dispersion σ_g for chemical transformation, as per Table 11.5; this is shown in the second line of Table 11.6. The third line shows the corresponding ϕ according to Eq.(11.31). Line four shows the ratio mean/median for v_{dep}, also according to Eq.(11.31). The total σ_g of Table 11.5 is listed in line five of Table 11.6, with the corresponding ϕ in line six. Line seven ("mean/median, total uncorrected") shows the simple result of applying Eq. (11.31) directly. Since that result would be correct if v_{dep} were in the numerator rather than the denominator, we have to correct it by dividing by the square of the mean/median ratio of $1/v_{dep}$, i.e. by 1.09^2 for PM. That yields the numbers for "mean/median, total corrected" in line eight. Line nine is the inverse of line eight. Finally the lower and upper ends of the confidence intervals are shown in lines ten and eleven. For PM the ratio lower bound/mean is 0.73/2.65 = 0.28 and the upper bound/mean is 0.73 · 2.65 = 1.94.

11.6 Presentation of Uncertainty

Communicating the uncertainties of external costs is very important, to ensure that users understand the limitations. Since 1998 ExternE has made a concerted effort to show the uncertainties. Unfortunately it is much more difficult to deal with both a num-

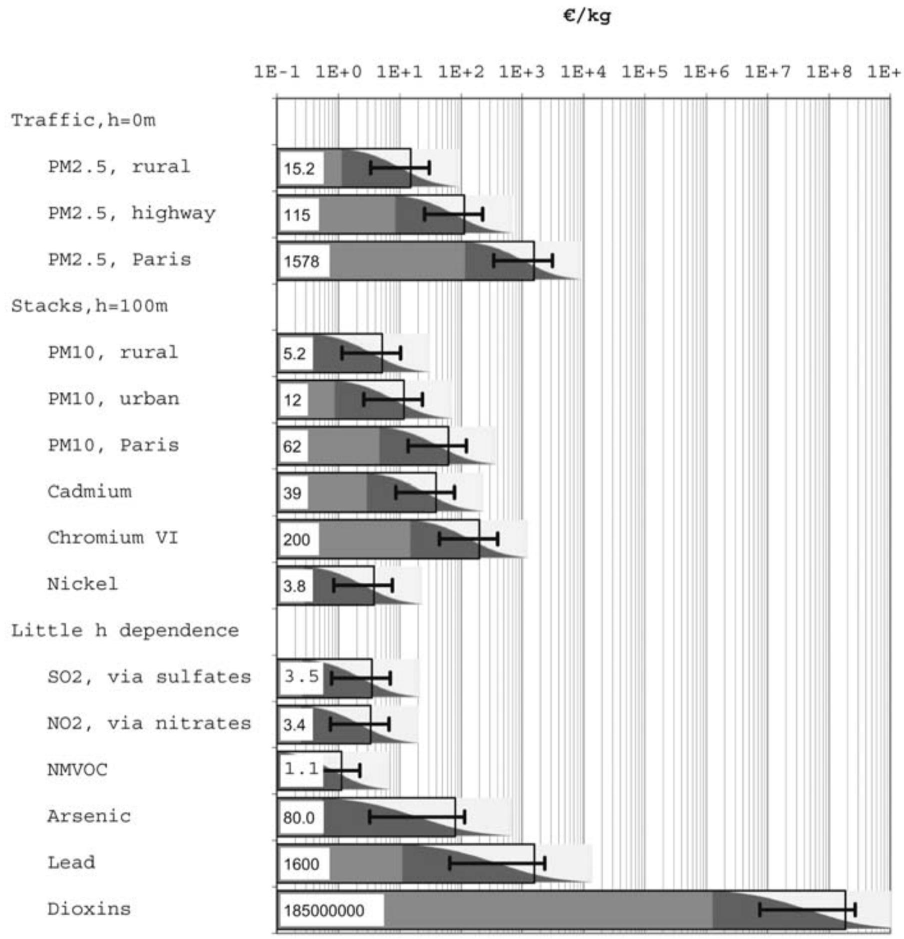

Figure 11.7 A possible format for representing the uncertainty of damage costs. The numbers are for LCA applications in the EU15 (Rabl *et al.*, 2004). The error bars indicate the 68% confidence interval; on the logarithmic scale they are symmetric around the median (= geometric mean of lognormal distribution). Broad white bars and numbers are the mean which is larger than the median. The gray S-shaped curve indicates the probability that the true cost is above a specified value.

ber and its uncertainty than just with the number. Several formats have been tried, for example giving a high and a low estimate in addition to the central estimate. In some cases the sensitivity to certain assumptions was also shown, for instance the use of VPF or of VOLY for mortality valuation in the 1998 reports. However, doing so complicates the presentation of the results, with an awkward proliferation of numbers if more than a few sensitivity studies are shown. Often users focus on just the central estimate without paying attention to the uncertainties, no matter how clearly they are displayed. To prevent readers from doing so, European Commission (1999) showed the global warming costs (whose uncertainty is notoriously large) as a range rather than a single number. That does not seem to be a good approach because many users extracted a single number by taking an average – not appropriate since the probability distribution is lognormal and the average is much larger than the correct number, namely the geometric mean.

Noting that a perfect representation of the results for each and every user is not possible, we present in Figure 11.7 a possible format which contains a great deal of information in a fairly concise and instructive form, by indicating not only the damage cost with its 68% confidence interval (as error bars) but also the cumulative probability distribution (as S-shaped curve). For PM, NO_x and SO_2 we have approximated σ_g as 3 and for As, Pb and dioxins as 6 (larger because of the ingestion pathways).

11.7 Evolution of Damage Cost Estimates over Time

Damage cost estimates change with time because of scientific progress. During the last decade there has been intense world-wide research in air pollution epidemiology, so changes in CRFs should not come as a surprise. In addition there have been major changes in the monetary valuation of air pollution mortality. The resulting evolution of damage cost estimates by ExternE is shown in Figure 11.8. These changes are compatible with the confidence intervals we have estimated.

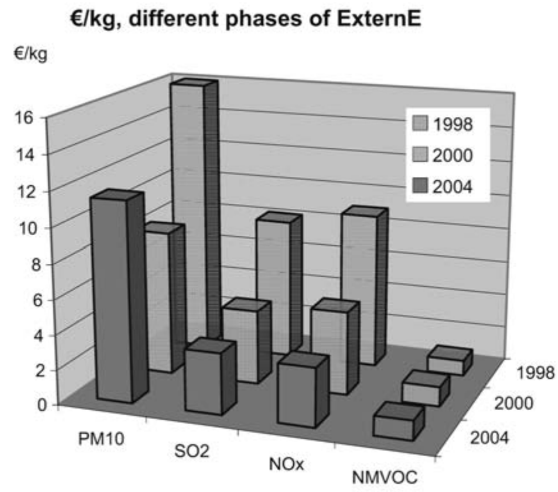

Figure 11.8 Damage cost estimates, for LCA applications in EU15, by different phases of ExternE.

11.8 Consequences of the Uncertainties for Decisions

Since many people have questioned the usefulness of the ExternE results because of their large uncertainty, we emphasise that the uncertainties should not purely be looked at by themselves; rather one should ask what effect the uncertainties have on the choice of policy options. The key question to be asked is "how large is the cost penalty if one makes the wrong choice because of errors or uncertainties in the cost or benefit estimates?" In a recent paper Rabl, Spadaro and van der Zwaan (2005) have looked at the uncertainties from this perspective and their findings are very encouraging: the risk of cost penalties is surprisingly small even with the very large uncertainties of ExternE.

It is instructive to distinguish between policy decisions that are binary (e.g. choice between nuclear or coal-fired power plant) or continuous (e.g. what limit to set for the SO_2 emissions from a power plant). For binary decisions the situation is sometimes quite simple because the uncertainty, even if very large, has no effect if it does not change the ranking. For example, in France the market cost of nuclear is lower than that of coal for baseload and that ranking does not change if external costs are included; furthermore, the external costs of nuclear are so much lower than those of coal that the ranking is not affected by the uncertainties: even if one were to take the 1 σ_g upper limit for nuclear and the 1 σ_g lower limit for coal, the ranking would still remain the same. A similar conclusion was reached in a recent cost-benefit analysis of the proposed new regulations for emissions of particles and SO_2 from incinerators:

compared to the old regulations the new ones are justified, even if one takes the 1 σ_g lower limit for the damage costs.

For continuous choices the effect of uncertainty is surprisingly small because, near an optimum, the total social cost varies only slowly as individual cost components are varied. Specifically, using abatement cost curves for NO_x, SO_2, dioxins and CO_2, Rabl, Spadaro and van der Zwaan (2005) evaluate the cost penalty for errors in the following choices: national emission ceilings for NO_x and SO_2 in each of 12 countries of Europe, an emission ceiling for dioxins in the UK, and limits for the emission of CO_2 in Europe. As an example, Figure 11.9 shows the cost penalty ratio *R* for national emission ceilings for SO_2 if a wrong level is chosen because of an error in the damage cost estimate. The graph is non-dimensional, *R* being the relative increase in total social cost (abatement cost plus damage cost) above the true optimum and $x = D_{true}/D_{est}$ the error in the damage cost estimate.

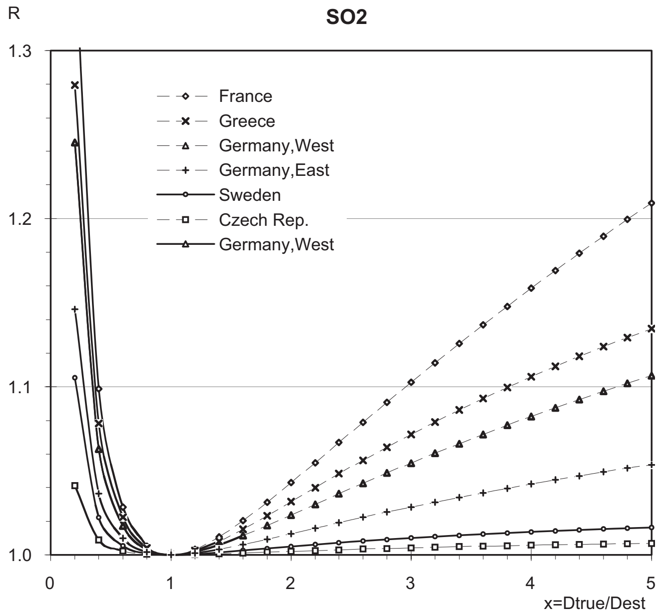

Figure 11.9 The cost penalty ratio R versus the error $x = D_{true}/D_{est}$ in the damage cost estimate for several countries, selected to show extremes as well as intermediate curves. The labels are placed in the same order as the curves. Dashed lines correspond to extrapolated regions of the cost curves. From Rabl, Spadaro and van der Zwaan (2005).

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Appendix: Glossary and Conversion Factors

As	arsenic
BS	black smoke
c	concentration
C	damage cost [€/kg of pollutant]
$c_{air}(x,q)$	increase in concentration [$\mu g/m^3$] at a point x = (x,y) due to the emission q
CB	chronic bronchitis
CO	carbon monoxide
COPD	chronic obstructive pulmonary disease
Cr VI	chromium in oxidation state 6
CRF	concentration-response function
D	$\dot{I}_{inhal}(\dot{m}) / \dot{m}$ = damage [damage units/kg of pollutant]
DRF	dose-response function
D_{uni}	damage calculated by UWM
E	exposure per emitted quantity of pollutant [receptors·($\mu g/m^3$)/($\mu g/s$)]
EC	European Commission
EPA	Environmental Protection Agency of USA
$F_{dep}(\mathbf{x},q)$	deposition flux [$\mu g/(m^2 \cdot s)$]
f_{pop}	fraction of the population affected by the end point in question.
HA	hospital admission
Hg	mercury
I(q)	impact rate [cases/yr], and
I_{ref}.	baseline or reference level of incidence of the end point in question.
LE	life expectancy
LRS	lower respiratory symptoms
mRAD	minor restricted activity day
N	nitrogen
Ni	nickel
NMVOC	non-methane volatile organic compounds
NO_x	unspecified mixture of NO and NO_2
O_3	ozone
OR	odds ratio = output of case-control studies; in the limit of small risks (relevant for most air pollution impacts) the OR becomes equal to the RR
p	unit cost [€/case]
PAH	polycyclic aromatic hydrocarbons
Pb	lead

PM	particulate matter
PM_d	particulate matter with aerodynamic diameter smaller than d μm
ppb	parts per billion
q	emission rate of pollutant [kg/s],
RAD	restricted activity day
RR	relative risk
S	sulphur
s_{CR}	slope of CR function [cases/(person·yr·μg/m^3)]
$s_{CR}(x)$	slope of CRF at x [(cases/yr)/(receptor·(μg/m^3))].
TEQ	toxic equivalence 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD)
URF	unit risk factor = probability that a person of standard weight of 70 kg will develop cancer due to exposure (by inhalation) to a concentration of 1 μg/m^3 of a pollutant over a 70-year lifetime.
URS	upper respiratory symptoms
UWM	uniform world model
v_{dep}	deposition velocity [m/s]
$v_{dep,eff}$	effective deposition velocity, including chemical transformation [m/s]
VOC	volatile organic compounds
VOLY	value of a life year (value of a YOLL)
VPF	value of a prevented fatality = VSL
VSL	value of statistical life = VPF
WDL	work day lost
WHO	World Health Organization
WTP	willingness-to-pay
YOLL	years of life lost
ϕ	$\ln(\sigma_g)$
μ	mean
μ_g	geometric mean
$\rho(x)$	density of receptors (population, buildings, crops, etc.) [receptors/m^2] at x
σ	standard deviation
σ_g	geometric standard deviation
ξ	$\ln(\mu_g)$

1 ppb O_3 = 1.997 μg/m^3 of O_3
1 ppb NO_2 = 1.913 μg/m^3 of NO_2
1 ppm CO = 1.165 mg/m^3 of CO

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