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ExternE

Externalities of Energy

METHODOLOGY ANNEXES

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Introduction

The project has attempted to quantify the external costs and benefits of the major electricity generation technologies in Europe, and to aggregate these damages for national power systems, and to apply results to policy making issues. Therefore, representative technologies have been selected for the participant countries, based on the existing power systems, or on the expected development of these systems.

The methodology used for the assessment of the externalities of the fuel cycles selected has been the one developed within the ExternE Project. This is a bottom-up methodology, which uses the “impact pathway” approach. Emissions and other types of burden such as risk of accident are quantified and followed through to impact assessment and valuation. The approach thus provides a logical way of quantifying externalities.

The underlying principles on which the methodology has been developed are transparency, consistency, and comprehensiveness of the analysis. These characteristics should be present along the stages of the methodology, namely: site and technology characterisation, identification of burdens and impacts, prioritisation of impacts, quantification, and economic valuation.

More details on the methodology in general, and on the specific methods for the valuation of each impact, may be found in the report issued by the ExternE Core Project (European Commission, 1998a), within which the methodology has been updated and further developed.

Here some Annexes explaining some of the methodological issues as well as EcoSense, the tool used, are given.

ANNEXES

Appendix I: The Ecosense model

APPENDIX I: THE ECOSENSE MODEL

1. Introduction

Since the increasing understanding of the major importance of long range transboundary transport of airborne pollutants also in the context of external costs from electricity generation, there was an obvious need for a harmonised European-wide database supporting the assessment of environmental impacts from air pollution. In the very beginning of the ExternE Project, work was focused on the assessment of local scale impacts, and teams from different countries made use of the data sources available in each country. Although many teams spent a considerable amount of time compiling data on e.g. population distribution, land use etc., we had to realise that country specific data sources and grid systems were hardly compatible when we had to extend our analysis to the European scale. So it was logical to set up a common European-wide database by using official sources like EUROSTAT and make it available to all ExternE teams. Once we had a common database, the consequent 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.

Taking into account this background, the objectives for the development of the EcoSense model 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 into the system. As a consequence, all the calculation modules (except for the ISC-model, see below) are designed in a way that they are a *model-interpreter* rather than a *model*. Model specifications like e. g. chemical equations, dose-response functions or monetary values are stored in the database and can be modified by the user. This concept allows an easy modification of model parameters, and at the same time the model does not necessarily appear as a black box, as the user can trace back what the system is actually doing.

2. 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 impacts is based on the evaluation of statistics rather than on modelling. Version 2.0 of EcoSense covers 13 pollutants, including the ‘classical’ pollutants SO_2, NO_x, particulates and CO, as well as some of the most important heavy metals and hydrocarbons, but does not include impacts from radioactive nuclides.

3. The EcoSense Modules

Figure 1 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 models, which are 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. The following sections give a more detailed description of the different EcoSense modules.

3.1 The EcoSense database

3.1.1 Reference Technology Database

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

3.1.2 Reference Environment Database

The reference environment database is the core element of the EcoSense database, providing data on the distribution of receptors, meteorology as well as a European wide emission inventory. All geographical information is organised using the EUROGRID co-ordinate

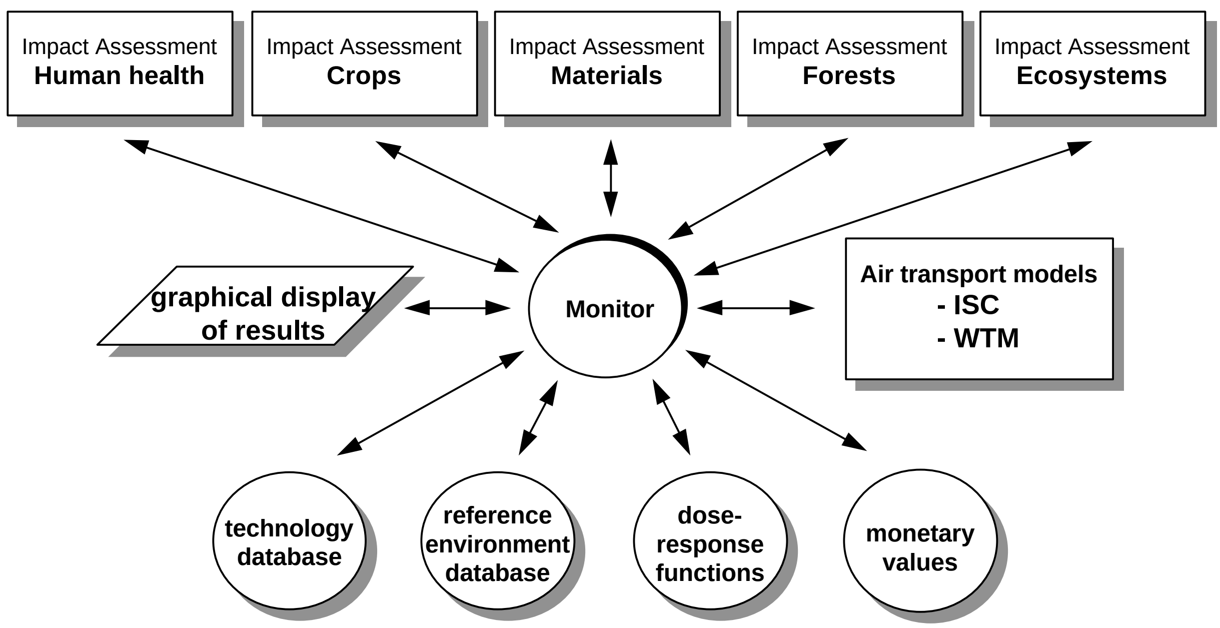

Figure 1 Structure of the EcoSense model

system, which defines equal-area projection gridcells of 10 000 km^2 and 100 km^2 (Bonnefous and Despres, 1989), covering all EU and European non-EU countries.

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

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

3.1.3 Exposure-Response Functions

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

3.1.4 Monetary Values

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

3.2 Air Quality Models

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

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

All input data required to run the Windrose Trajectory Model are provided by the EcoSense database. A set of site specific meteorological data has to be added by the user to perform local scale modelling using the ISC model. The concentration and deposition fields calculated by the air quality models are stored in the reference environment database. Section 4 gives a more detailed description of the two models.

3.3 Impact Assessment Modules

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

3.4 Presentation of Results

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

4. The Air Quality Models Integrated in EcoSense

4.1 Local scale modelling of primary pollutants - the Industrial Source Complex model

Close to the plant, i.e. at distances of some 10-50 km from the plant, chemical reactions in the atmosphere have little influence on the concentrations of primary pollutants, if NO and its oxidised counterpart NO_2 can be summarised as NO_x. Due to the large emission height on top of a tall stack, the near surface ambient concentrations of the pollutants at short distances from the stack are heavily dependent on the vertical mixing of the lower atmosphere. Vertical mixing depends on the atmospheric stability and the existence and height of inversion layers (whether below or above the plume). For these reasons, the most economic way of assessing ambient air concentrations of primary pollutants on a local scale is a model which neglects chemical reactions but is detailed enough in the description of turbulent diffusion and vertical mixing.

An often used model which meets these requirements is the Gaussian plume model. The concentration distribution from a continuous release into the atmosphere is assumed to have a Gaussian shape:

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

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

s_y	standard deviation of lateral concentration distribution at downwind distance x
s_z	standard deviation of vertical concentration distribution at downwind distance x
h	plume height above terrain

The assumptions embodied into this type of model include those of idealised terrain and meteorological conditions so that the plume travels with the wind in a straight line. Dynamic features which 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 rather justified for a statistical evaluation of a long period, where mutual changes in wind direction cancel out each other, 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 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 of depressed terrain.
- The mixing height is terrain following.
- The effective plume stabilisation height h_{stab} at receptor location (x,y) is given by:

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

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

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

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

4.2 Regional scale modelling of primary pollutants and acid deposition - the Windrose Trajectory Model

With increasing distance from the stack the plume spreads vertically and horizontally due to atmospheric turbulence. Outside the area of the local analysis (i.e. at distances beyond 50 km from the stack), it can be assumed for most purposes that the pollutants have vertically been mixed throughout the height of the mixing layer of the atmosphere. On the other hand, chemical transformations can no longer be neglected on a regional scale. The most economic way to assess annual, regional scale pollution is a model with a simple representation of transport and a detailed enough representation of chemical reactions.

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 2.

In EcoSense, the model is implemented by means of

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

The 1990 meteorological data were provided by the Meteorological Synthesizing Centre-West of EMEP at The Norwegian Meteorological Institute (Hollingsworth, 1987), (Nordeng, 1986). 6-hourly data in the EMEP 150 km grid of precipitation and wind (at the 925 hPa level) were transformed to the EUROGRID grid and averaged to obtain, receptor specific, the mean annual wind rose (frequency distribution of the wind per sector), the mean annual windspeed, and total annual precipitation. Base line emissions of NO_x, SO_2 and NH_3 for Europe are taken from the 1990 EMEP inventory (Sandnes and Styve, 1992).

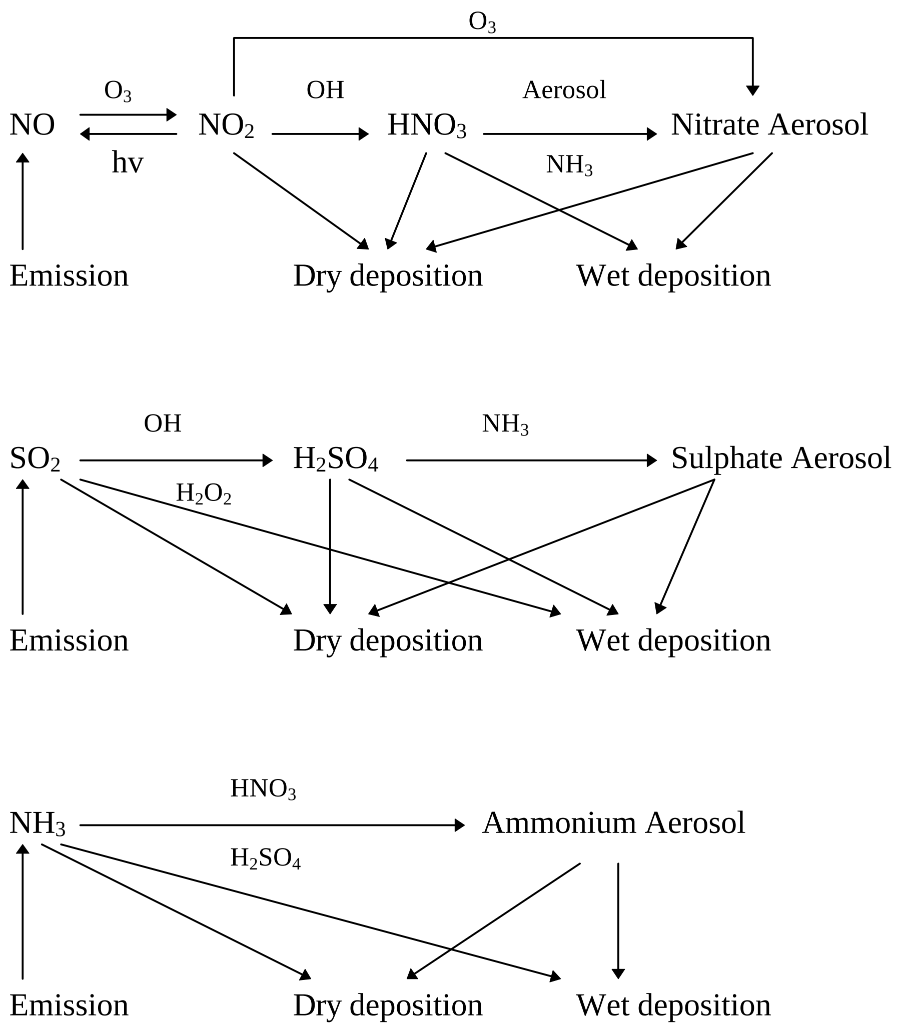

Figure 2 Chemical scheme in WTM, adopted from Derwent *et al* (1993)

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APPENDIX II: ANALYSIS OF HEALTH EFFECTS

1. Introduction

Five types of health effect have been dealt with in the present study;
1. Non-carcinogenic effects of air pollutants
2. Carcinogenic effects of radionuclide emissions
3. Carcinogenic effects of dioxins and trace metals
4. Occupational health issues (disease and accidents)
5. Accidents affecting members of the public

Each of these is discussed briefly below. A more complete description of the assumptions made is given in the ExternE methodology report (European Commission, 1998), and for carcinogenic effects of radionuclides in the earlier report on the nuclear fuel cycle (European Commission, 1995a).

2. Non-Carcinogenic Effects of Air Pollutants

2.1 Introduction

Within ExternE this category of impact has mainly dealt with the following primary and secondary pollutants, in relation to analysis of the effects of power stations.

NO_x	SO_2	NH_3	CO
ozone	nitrate aerosol	sulphate aerosol	PM_x

Other pollutants could be added to the list but early analysis (European Commission, 1995b, p. 93; based on Maier *et al*, 1992) suggested that the amounts emitted from power stations would be negligible. A possible exception concerned mercury, whose high volatility results in poor capture by flue gas scrubbing equipment.

2.2 Epidemiological evidence

The available literature on the pollutants listed has been reviewed by Hurley, Donnan and their colleagues, providing the exposure-response functions listed in Tables 1 and 2. Further details on the uncertainty classification given in the final column of the table are given in Appendix VIII. The uncertainty rating provides an assessment of uncertainty throughout the chain of analysis - in other words from quantification of emissions through to valuation of damage. Table 1 contains the 'core' set of exposure-response functions used in ExternE. Table 2 contains functions recommended only for use in sensitivity analysis.

Table 1: Quantification of human health impacts. The exposure response slope, f_{er} , is for Western Europe and has units of [cases/(yr-person-µg/m^3)] for morbidity, and [%change in annual mortality rate/(µg/m^3)] for mortality.

Receptor	Impact Category	Reference	Pollutant	f_{er} [1]	Uncertainty rating
ASTHMATICS (3.5% of population)					
adults	Bronchodilator usage	Dusseldorp *et al*, 1995	PM_{10}, Nitrates, $PM_{2.5}$, Sulphates	0.163 0.163 0.272 0.272	B B? B B
	Cough	Dusseldorp *et al*, 1995	PM_{10}, Nitrates, $PM_{2.5}$, Sulphates	0.168 0.168 0.280 0.280	A A? A A
	Lower respiratory symptoms (wheeze)	Dusseldorp *et al*, 1995	PM_{10}, Nitrates, $PM_{2.5}$, Sulphates	0.061 0.061 0.101 0.101	A A? A A
children	Bronchodilator usage	Roemer *et al*, 1993	PM_{10}, Nitrates, $PM_{2.5}$, Sulphates	0.078 0.078 0.129 0.129	B B? B B
	Cough	Pope and Dockery, 1992	PM_{10}, Nitrates, $PM_{2.5}$, Sulphates	0.133 0.133 0.223 0.223	A A? A A
	Lower respiratory symptoms (wheeze)	Roemer *et al*, 1993	PM_{10}, Nitrates, $PM_{2.5}$, Sulphates	0.103 0.103 0.172 0.172	A A? A A
all	Asthma attacks (AA)	Whittemore and Korn, 1980	O_3	4.29E-3	B?
ELDERLY 65+ (14% of population)					
	Congestive heart failure	Schwartz and Morris, 1995	PM_{10}, Nitrates, $PM_{2.5}$, Sulphates, CO	1.85E-5 1.85E-5 3.09E-5 3.09E-5 5.55E-7	B B? B B B
CHILDREN (20% of population)					
	Chronic bronchitis	Dockery *et al*, 1989	PM_{10}, Nitrates, $PM_{2.5}$, Sulphates	1.61E-3 1.61E-3 2.69E-3 2.69E-3	B B? B B
	Chronic cough	Dockery *et al*, 1989	PM_{10}, Nitrates, $PM_{2.5}$, Sulphates	2.07E-3 2.07E-3 3.46E-3 3.46E-3	B B? B B

1 Sources: [ExternE, European Commission, 1995] and [Hurley *et al*, 1997].

Table 1 (continued)**:** Quantification of human health impacts.

Receptor	Impact Category	Reference	Pollutant	fer	Uncertainty rating
ADULTS (80% of population)					
	Restricted activity days (RAD)[2]	Ostro, 1987	PM_{10},	0.025	B
			Nitrates,	0.025	B?
			$PM_{2.5}$,	0.042	B
			Sulphates	0.042	B
	Minor restricted activity day (MRAD)[3]	Ostro and Rothschild, 1989	O_3	9.76E-3	B
	Chronic bronchitis	Abbey *et al*, 1995	PM_{10},	4.9E-5	A
			Nitrates,	4.9E-5	A?
			$PM_{2.5}$,	7.8E-5	A
			Sulphates	7.8E-5	A
ENTIRE POPULATION					
	Respiratory hospital admissions (RHA)	Dab *et al*, 1996	PM_{10},	2.07E-6	A
			Nitrates,	2.07E-6	A?
			$PM_{2.5}$,	3.46E-6	A
			Sulphates	3.46E-6	A
		Ponce de Leon, 1996	SO_2	2.04E-6	A
			O_3	7.09E-6	A
	Cerebrovascular hospital admissions	Wordley *et al*, 1997	PM_{10},	5.04E-6	B
			Nitrates,	5.04E-6	B?
			$PM_{2.5}$,	8.42E-6	B
			Sulphates	8.42E-6	B
	Symptom days	Krupnick *et al*, 1990	O_3	0.033	A
	Cancer risk estimates	Pilkington and Hurley, 1997	Benzene	1.14E-7	A
			Benzo[a]Pyrene	1.43E-3	A
			1,3 butadiene	4.29E-6	A
			Diesel particles	4.86E-7	A
	Acute Mortality (AM)	Spix and Wichmann, 1996; Verhoeff *et al*, 1996	PM_{10},	0.040%	B
			Nitrates,	0.040%	B?
			$PM_{2.5}$,	0.068%	B
			Sulphates	0.068%	B
		Anderson *et al*, 1996, Touloumi *et al*, 1996	SO_2	0.072%	B
		Sunyer *et al*, 1996	O_3	0.059%	B
	Chronic Mortality (CM)	Pope *et al*, 1995	PM_{10},	0.39%	B
			Nitrates,	0.39%	B?
			$PM_{2.5}$,	0.64%	B
			Sulphates	0.64%	B

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

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

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

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

Table 2: Human health E-R functions for *sensitivity analysis only* (Western Europe). The exposure response slope, f_{er} , is for Western Europe and has units of [cases/(yr-person-µg/m^3)] for morbidity, and [%change in annual mortality rate/(µg/m^3)] for mortality.

Receptor	Impact Category	Reference	Pollutant	fer[1]	Uncertainty rating
ELDERLY, 65+ (14% of population)					
	Ischaemic heart disease	Schwartz and Morris, 1995	PM_{10},	1.75E-5	B
			Nitrates,	1.75E-5	B?
			$PM_{2.5}$,	2.92E-5	B
			Sulphates	2.92E-5	B
			CO	4.17E-7	B
ENTIRE POPULATION					
	Respiratory hospital admissions (RHA)	Ponce de Leon, 1996	NO_2	2.34E-6	A?
	ERV for COPD	Sunyer *et al*, 1993	Nitrates, PM_{10}	7.20E-6	B?
			Sulphates, $PM_{2.5}$	1.20E-5	B?
	ERV for asthma	Schwartz, 1993 and Bates *et al*, 1990	Nitrates, PM_{10}	6.45E-6	B?
			Sulphates, $PM_{2.5}$	1.08E-5	B?
		Cody *et al*, 1992 and Bates *et al*, 1990	O_3	1.32E-5	B?
	ERV for croup in pre school children	Schwartz *et al*, 1991	Nitrates, PM_{10}	2.91E-5	B?
			Sulphates, $PM_{2.5}$	4.86E-5	B?
	Cancer risk estimates	Pilkington and Hurley, 1997	Formaldehyde	1.43E-7	B?
	Acute Mortality (AM)	Touloumi *et al*, 1994	CO	0.0015%	B?
		Sunyer *et al*, 1996, Anderson *et al*, 1996	NO_2	0.034%	B?

[1] Sources: [EC, 1995c] and [Hurley and Donnan, 1997].

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

The main problem with interpretation of epidemiological data relates to covariation in parameters. This is particularly the case when seeking to ascribe blame between different pollutants, on the grounds that most of them are released simultaneously from similar sources. This creates a danger of double counting damages (essentially by attributing the same cases of whatever type of health effect to two or more pollutants). Much care has therefore gone into the selection of functions in this study to ensure so far as possible that this is avoided.

The epidemiological literature, in the context of other evidence, was reviewed to form a position on:

a) What ambient air pollutants have been shown as *associated* with adverse health effects (acute or chronic), and for what specific endpoints;
b) Which of these associations may reasonably be interpreted as *causal*; (It is important in assessing the effect of *incremental* pollution in ExternE to quantify *causal* relationships, and not just epidemiological associations).

c) What studies provide a basis for a good set of E-R functions, for quantifying the public health effects of incremental air pollution; and
d) How if at all should the E-R functions from individual studies be adapted for use in ExternE.

Judgements at all of these stages are the focus of debate currently among scientists and policy makers concerned with the health effects of air pollution. The most important issues are listed below, but see also the more thorough discussion provided by Hurley and Donnan (European Commission, 1998).

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

The link between particulates and health effects is now well accepted, even if the mechanisms for various effects remain elusive. Much debate was given to the best way of representing particles within the analysis. This needed to take account of the size of particles and their chemical characteristics. It was recommended that for the main implementation particles be described on a unit mass basis, and that E-R functions for particles should be indexed differently according to the source, as follows:

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

There is also good evidence from the APHEA study in Europe that ozone causes health effects, and that these are additive to those of particulates. To fit with available data on ozone levels, functions are expressed relative to the average of daily peak 6 hourly ozone concentrations.

In the last phase of ExternE we concluded that the evidence for SO_2 damaging health was too weak for functions to be recommended. However, in the APHEA studies, the size of the apparent SO_2 effect did not depend on the background concentrations of ambient particles. In the context of the evidence as a whole, including this result, it is recommended that the functions for SO_2 are used in the main ExternE implementations now; and that the estimated impacts are added to the effects of particles and of ozone.

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

a) the functions for CO and acute hospital admissions for congestive heart failure are used;

b) the functions for CO and acute mortality are not used.
Sensitivity analyses should consider including both, or omitting both.

In ExternE 1995, the epidemiological evidence regarding NO_2 was assessed. Some studies reported NO_2 effects. However, the broad thrust of the evidence then was that apparent NO_2 effects were best understood not as causal, but as NO_2 being a surrogate for some mixture of (traffic-related) pollution. It was concluded that a direct effect of NO_2 should not be quantified, though indirectly, NO_x did contribute, as a precursor to nitrates and to ozone. Review of the APHEA study results led to the same conclusion. Thus for the main analyses, the E-R relationships for NO_2 are not used, though they can be applied in the sensitivity analyses.

For many of these pollutants, there clearly is a threshold *at the individual level*, in the sense that most people are not realistically at risk of severe acute health effects at current background levels of air pollution. There is however no good evidence of a threshold *at the population level*; i.e. it appears that, for a large population even at low background concentrations, some vulnerable people are exposed some of the time to concentrations which do have an adverse effect. This understanding first grew in the context of ambient particles, where the ‘no threshold’ concept is now quite well established as a basis for understanding and for policy.

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

The final main issue concerns transferability of functions from the place in which data is collected. Differences have been noted in the course of this study between functions reported in different parts of Europe, and between functions derived in Europe compared to those from the USA. For the present work functions representative of cities in western Europe have been selected wherever possible (western Europe providing the focus for the analysis). Some functions have been brought in from US studies. Comparison of available data on similar end-points has allowed the use of scaling factors in transferring North American data to Europe. The use of such factors is not without controversy, and the selection of scaling factors somewhat arbitrary. However, the alternative, not to correct, implies a scaling factor of 1, which available evidence suggests is wrong.

2.3 Valuation

The Value of a Statistical Life (VOSL) of 2.6 MECU used in earlier ExternE reports has been adjusted to 3.1 MECU, to bring it into line with January 1995 prices. However, in earlier phases of the project a number of questions were raised regarding the use of the VOSL for every case of mortality considered. These originally related to the fact that many people whose deaths were linked to air pollution were suspected of having only a short life expectancy even in the absence of air pollution. Is it therefore logical to ascribe the same value to someone with a day to live as someone with tens of years of remaining life expectancy? Furthermore, is it logical to ascribe the full VOSL to cases where air pollution is only one factor of perhaps several that determines the time of death? In view of this the ExternE Project team has explored valuation on the basis of life years lost. For quantification of the value of a life year (VOLY) it has been necessary to adapt our estimate of the VOSL. This is not ideal by any means, but it does provide a first estimate for the VOLY.

A valid criticism of the VOLY approach is that people responding to risk seem unlikely to structure their response from some sense of their remaining life expectancy. It has been noted that the VOSL does not decline anything like as rapidly with age as would be expected if this were the case. However, one of the main reasons for this appears to be that a major component of the VOSL is attributable to a 'fear of dying'. Given that death is inevitable, there is no way that policy makers can affect this part of the VOSL. They can, however, affect the life expectancy of the population, leading back to assessment based on life years lost.

Within ExternE it has been concluded that VOSL estimates should be restricted to valuing fatal accidents, mortality impacts in climate change modelling and similar cases where the impact is sudden and where the affected population is similar to the general population. VOSL should not be used in cases where the hazard has a significant latency period before impact, or where the probability of survival after impact is altered over a prolonged period. In such cases the value of life years (VOLY) lost approach is recommended. However, in view of the continuing debate in this area among experienced and respected practitioners, VOSL is retained for sensitivity analysis.

The VOLY approach is particularly recommended for deaths arising from illnesses linked to exposure to air pollution. The value will depend on a number of factors, such as how long it takes for the exposure to result in the illness and how long a survival period the individual has after contracting the disease. On the basis of the best data available at the time, two sets of values have been estimated for PM impacts: one for acute mortality and for chronic mortality. Both sets will vary according to the discount rate. Estimated values for acute mortality are: 73,500 ECU (0% discount rate), 116,250 (3% discount rate) and 234,000 ECU (10% discount rate). For chronic mortality the corresponding values are: 98,000 ECU (0% discount rate), 84,330 ECU (3% discount rate), and 60,340 ECU (10% discount rate).

Updated values for morbidity effects are given in Table AII.?. Compared to the earlier report (European Commission, 1995) most differences reflect an inflation factor, but some new effects are included, notably chronic bronchitis, chronic asthma, and change in prevalence of cough in children.

Table AII.? Updated values in ECU for morbidity impacts.

Endpoint	New Value	Estimation Method and Comments
Acute Morbidity		
Restricted Activity Day (RAD)	75	CVM in US estimating WTP. Inflation adjustment made.
Symptom Day (SD) and Minor Restricted Activity Day	7.5	CVM in US estimating WTP. Account has been taken of Navrud's study, and inflation.
Chest Discomfort Day or Acute Effect in Asthmatics (Wheeze)	7.5	CVM in US estimating WTP. Same value applies to children and adults. Inflation adjustment made.
Emergency Room Visits (ERV)	223	CVM in US estimating WTP. Inflation adjustment made.

Respiratory Hospital Admissions (RHA)	7,870	CVM in US estimating WTP. Inflation adjustment made.
Cardiovascular Hospital Admissions	7,870	As above. Inflation adjustment made.
Acute Asthma Attack	37	COI (adjusted to allow for difference between COI and WTP). Applies to both children and adults. Inflation adjustment made.
Chronic Morbidity		
Chronic Illness (VSC)	1,200,000	CVM in US estimating WTP. Inflation adjustment made
Chronic Bronchitis in Adults	105,000	Rowe et al (1995).
Non fatal Cancer	450,000	US study revised for inflation.
Malginant Neoplasms	450,000	AM suggested valuing as non-fatal cancer.
Chronic Case of Asthma	105,000	Based on treating chronic asthma as new cases of chronic bronchitis.
Cases of change in prevalance of bronchitis in children	225	Treated as cases of acute bronchitis.
Cases of change in prevalance of cough in children	225	As above.

3. Carcinogenic Effects of Radionuclide Emissions

3.1 Introduction

A brief explanation of the terminology specific to the nuclear fuel cycle assessment is presented in Box AI.1. Unlike the macropollutants described in the previous section, analysis of the effects of emissions of radionuclides is not carried out using the EcoSense model (it was not felt necessary, or practicable, to include every impact pathway for fuel chain analysis within EcoSense). In view of this it is necessary to give additional details of the methodology for assessment of the damages resulting from radionuclide emissions, compared to the information given in the other sections in this Appendix.

For the radiological impacts to the public and environment, independent evaluations must be done for each radionuclide in each mode of radionuclide release or exposure. The pathway analysis methodology presented by a CEC DGXII project for the assessment of radiological impact of routine releases of radionuclides to the environment (NRPB, 1994) has been used. Different models were required to evaluate the impact of accidents.

Box AII.1 Definitions

Becquerel - the basic unit of radioactivity.
(1 Bq = 1 disintegration per second = 2.7E-11 Ci) **(Bq).**

Absorbed Dose - is the fundamental dosimetric quantity in radiological protection. It is the energy absorbed per unit mass of the irradiated material. This is measured in the unit gray **(Gy)** (1 Gy = 1 joule/kg).

Dose Equivalent - is the weighted absorbed dose, taking into account the type and energy of the radiation. This is reported in the units of joule/kg with the name sievert (**Sv**) (1 Sv = 100 rem).
[**mSv** = 10^{-3} Sv].

Effective Dose - the weighted sum of the dose equivalents to the most sensitive organs and tissues **(Sv)**.

Committed Effective Dose - the effective dose integrated over 50 years for an adult. If doses to children are considered it is integrated over 70 years **(Sv)**.

Average Individual Dose - this term is used in this report as the committed effective dose that the average individual would be expected to receive under the conditions being assessed **(Sv).**

Collective Dose - to relate the exposure to the exposed groups or populations, the average individual dose representative of the population is multiplied by the number of people in the group to be considered **(man.Sv)**.

Physical Half-life ($T_{1/2}$) - time it takes for half the atoms of a radionuclide to decay (seconds, minutes, days, or years).

Environmental or Effective Half-life ($T_{1/2}$) - time it takes for the activity of a radionuclide to decrease by half in a given component of the ecosystem (seconds, minutes, days, or years). This is due to environmental & biological transfer and the physical half-life of the nuclide.

The damage to the general population (collective dose) is calculated based on assumptions for average adult individuals in the population. Differences in age and sex have not been taken into account. It is assumed that the number of people and their habits remain the same during the time periods assessed.

Atmospheric, liquid and sub-surface terrestrial releases are treated as separate pathways. Due to the different physical and chemical characteristics of the radionuclides, each nuclide is modelled independently and an independent exposure of dose calculated. This approach allows for the summation of all doses before application of the dose response coefficients.

Occupational impacts, radiological and non-radiological can often be based on published personnel monitoring data and occupational accident statistics. There is typically no modelling done for this part of the evaluation.

The evaluation of severe reactor accidents are treated separately due to their probabilistic nature and the need to use a different type of atmospheric dispersion model (European

Commission, 1998), though the principles for quantification of impacts remain the same as described here. Differences arise at the valuation stage.

Priority pathways can be modelled in varying degrees of complexity taking into account the particular radionuclide released, the physico-chemical forms of the release, the site-specific characteristics, and receptor-specific dose and response estimates. With validated models of the transfer of radionuclides in the environment, many nuclide-specific parameters have been determined. Generalised values applicable to European ecosystems have also been developed in Europe (NRPB, 1994), US (Till and Meyer, 1983) and by international agencies (UNSCEAR, 1993, International Commission on Radiological Protection (ICRP23, ICRP60). Site-specific data are used for population, meteorology, agricultural production and water use.

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

3.2 Boundaries of the Assessment

The assessment of the nuclear fuel chain requires, like any other, the definition of time and space boundaries. The objectives of this project require consistency in approach between different fuel chains, which broadly require the analysis to be as comprehensive as possible. Due to the long half-life of some of the radionuclides, low-level doses will exist very far into the future. These low-level doses can add up to large damages when spread across many people and many years (assuming constant conditions). The validity of this type of modelling has been widely discussed. On one hand, there is a need to evaluate all the possible impacts if a complete assessment of the fuel cycle is to be made. On the other hand, the uncertainty of the models increases and the level of doses that are estimated fall into the range where there is no clear evidence of resulting radiological health effects. The evaluation was completed using the conservative assumptions that:

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

The meaningfulness of carrying the assessment for long periods of time is highly questionable. This very long time scale presents some problems in the direct comparison of the nuclear fuel cycle with the other fuel chains on two counts; for example, lack of evaluation of long term toxic effects of heavy metals and chemicals released or disposed of in other fuel cycles. The assessment of the impacts on different space scales is not as problematic. It has been shown that the distance at which the evaluation stops can have a large influence on the final costs. For these reasons, the impacts estimated for the nuclear fuel cycle are presented in a time and space matrix. This form of presentation of results ensures that all the important impacts have been assessed and allows comparison of results in the categories that are appropriate. It can also be made clear that the uncertainty of the results increases with the scope and generality of the assessment.

Short-term is considered to include immediate impacts, such as occupational injuries and accidents; medium-term includes the time period from 1 to 100 years and long-term from 100 to 100,000 years. The limit of 100,000 years is arbitrary, however the most significant part of the impacts have been included.

3.3 Impacts of atmospheric releases of radionuclides

The most important impact pathways for public health resulting from atmospheric releases are:
- inhalation and external exposure due to immersion from the radionuclides in the air,
- external exposure from ground deposition, and
- ingestion of contaminated food resulting from ground deposition.

These pathways are illustrated in Figure AII.1.

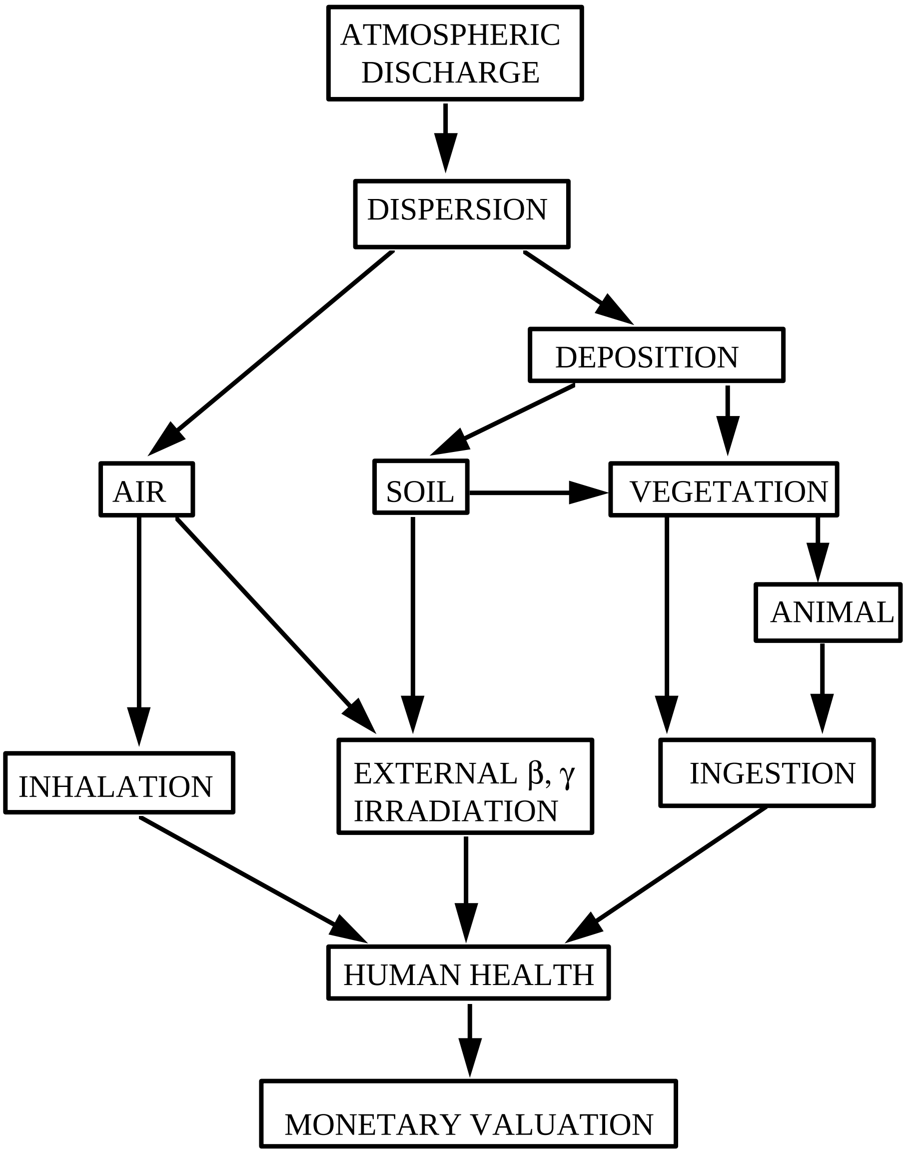

Figure AII.1 Impact pathway for an atmospheric release of radionuclides into the terrestrial environment

3.3.1 Dispersion

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

3.3.2 Exposure

Inhalation doses to the population occur at the first passage of the 'cloud' of radioactive material, and for the extremely long-lived, slow-depositing radionuclides (H-3, C-14, Kr-85, I-129), as they remain in the global air supply circulating the earth. Human exposure to them is estimated using the reference amount of air that is inhaled by the average adult (the 'standard reference man' (ICRP 23)), and nuclide-specific dose conversion factors for inhalation exposure in the local and regional areas (NRPB, 1991).

External exposure results from immersion in the cloud at the time of its passage and exposure to the radionuclides that deposit on the ground. The immediate exposure to the cloud passage is calculated for the local and regional areas. The global doses for exposure to the cloud are calculated for I-129 and Kr-85. For external exposure due to deposition, the exposure begins at the time of deposition but the length of time that must be included in the assessment depends on the rate of decay and rate of migration away from the ground surface. For example, as the radionuclide moves down in the soil column, the exposure of the population decreases due to lower exposure rates at the surface. The time spent out of doors will also affect the calculated dose because buildings act as shields to the exposure and therefore diminish the exposure. This is a case where the conservative assumption that the population spends all the time outside is taken.

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

A detailed environmental pathway model has not been used here. The environmental transfer factors between deposition and food concentration in different food categories, integrated over different time periods, assuming generalised European agricultural conditions was obtained from the NRPB agricultural pathway model FARMLAND. A constant annual deposition rate is assumed and the variation in the seasons of the year are not taken into account. The agricultural products are grouped, for this generalised methodology, as milk, beef, sheep, green vegetables, root vegetables and grains. Examples of the transfer factors used for a few radionuclides are given in Table AII.1.

Cultivated vegetation is either consumed directly by people or by the animals which ultimately provide milk and meat to the population. The exposures received by the population are calculated taking into consideration food preparation techniques and delay time between harvest and consumption to account from some loss of radioactivity. An average food consumption rate data (illustrated by the French data shown in Table AII.2) and population size is used for calculating the amount of food that is consumed in the local, regional and global population. The collective doses are calculated assuming that the food will be consumed locally but if there is an excess of agricultural production it will pass to the regional population next, and afterwards to the global population group. In this way the dose due to the total food supply produced within the 1000 km area included in the atmospheric dispersion assessment is taken into account.

Table AII.1 Food transfer coefficients, integrated over different time periods, for food products (in Bq/kg per Bq/m^2/s of deposition)

Products	Period (y)	I-129	I-131	Cs-137	U-238	Pu-239
Cow	30	1.85E+05	2.47E+04	9.14E+05	8.00E+03	4.53E+03
	50	1.98E+05	2.47E+04	9.14E+05	8.20E+03	4.54E+03
	100	2.09E+05	2.47E+04	9.14E+05	8.28E+03	4.54E+03
	200	2.11E+05	2.47E+04	9.14E+05	8.29E+03	4.54E+03
	100 000	2.13E+05	2.47E+04	9.14E+05	8.31E+03	4.54E+03
Green vegetables	30	1.69E+05	4.12E+04	1.42E+05	1.16E+05	1.05E+05
	50	1.90E+05	4.12E+04	1.45E+05	1.19E+05	1.05E+05
	100	2.31E+05	4.12E+04	1.47E+05	1.25E+05	1.05E+05
	200	3.22E+05	4.12E+04	1.48E+05	1.29E+05	1.05E+05
	100 000	3.29E+05	4.12E+04	1.48E+05	1.40E+05	1.05E+05
Root vegetables	30	1.83E+05	1.09E+04	1.56E+05	4.90E+03	9.29E+01
	50	2.05E+05	1.09E+04	1.59E+05	8.60E+03	1.46E+02
	100	2.46E+05	1.09E+04	1.62E+05	1.50E+04	2.51E+02
	200	2.70E+05	1.09E+04	1.63E+05	1.60E+04	3.10E+02
	100 000	3.44E+05	1.09E+04	1.63E+05	3.00E+04	5.02E+02
Milk	30	2.74E+05	5.82E+04	1.79E+05	2.42E+04	8.20E+01
	50	2.93E+05	5.82E+04	1.79E+05	2.48E+04	8.22E+01
	100	3.10E+05	5.82E+04	1.79E+05	2.50E+04	8.22E+01
	200	3.12E+05	5.82E+04	1.79E+05	2.51E+04	8.22E+01
	300	3.15E+05	5.82E+04	1.79E+05	2.51E+04	8.22E+01

Table AII.2 Average consumption rates for an average French adult.

Product	Consumption per year in kg
Cow	15
Sheep	2.7
Grain	53
Green vegetable	31
Root vegetable	48
Fresh milk	16
Other milk	69
Drinking water	550

3.3.3 Dose Assessment

It is possible to report a calculated dose by radionuclide, type of exposure and organ of the body, but for the purpose of estimating a population risk, a whole body effective collective dose was calculated taking into account these factors. A few examples of the dose conversion factors used in the evaluation are presented in Table AII.3.

The relationship between the dose received and the radiological health impact expected to result are based on the information included in the international recommendations of the ICRP60 (ICRP, 1990). The factors, or dose response functions, used to predict the expected occurrence of cancer over a lifetime or severe hereditary effects in future generations per unit exposure received by the general public are 0.05 fatal cancers per manSv (unit of collective dose) and 0.01 severe hereditary effects in future generations per manSv.

The fraction of cancers that would be expected to be non-fatal (0.12 non-fatal cancers per manSv) are calculated based on the expected number of fatal cancers and the lethality fractions reported for 9 categories of cancer in ICRP60. This is reflected in the aggregated non-fatal cancer factor of 0.12 per manSv.

It is recognised that the dose-response functions that are chosen in the assessment of radiological health effects are extremely important. There is still controversy on the exact values to use and different models have been proposed. Within the context of this project, internationally accepted factors have been used, assuming a linear response to radiation with no threshold, and a dose and dose rate effectiveness factor (DDREF) of 2. The DDREF is the factor used to extrapolate the data that exists for high-levels of exposure to the low levels of exposure of concern in this project. Detailed calculations were presented in the French analysis of the nuclear fuel chain under ExternE (European Commission, 1995a) in a way that allows the reader to apply different factors if desired.

Table AII.3 Dose conversion factors for exposure by ingestion and inhalation of radionuclides (Sv/Bq).

Radionuclide	Half-life	Type of release	Type of exposure	Dose conversion factor (Sv/Bq)
H-3	12.3 y	Liquid, gaseous	Ingestion	1.80 E-11
			Inhalation	1.73 E-11
C-14	5710 y	Liquid, gaseous	Ingestion	5.60 E-10
			Inhalation	5.60 E-10
I-129	1.6 E7 y	Gaseous	Ingestion	1.10 E-07
			Inhalation	6.70 E-08
I-131	8.1 d	Liquid, gaseous	Ingestion	2.20 E-08
			Inhalation	1.30 E-08
Cs-134	2.1 y	Liquid, gaseous	Ingestion	1.90 E-08
			Inhalation	1.20 E-08
Cs-137	30 y	Liquid, gaseous	Ingestion	1.30 E-08
			Inhalation	8.50 E-09
U-234	2.5 E5 y	Liquid, gaseous	Ingestion	3.90 E-08
			Inhalation	2.00 E-06
U-235	7.1E8 y	Liquid, gaseous	Ingestion	3.70 E-08
			Inhalation	1.80 E-06
U-238	4.5 E9 y	Liquid, gaseous	Ingestion	3.60 E-08

			Inhalation	1.90 E-06
Pu-238	86.4 y	Liquid, gaseous	Ingestion	2.60 E-07
			Inhalation	6.20 E-05
Pu-239	2.4 E4 y	Liquid, gaseous	Ingestion	2.80 E-07
			Inhalation	6.80 E-05

3.3.4 Time Distribution of the Expected Occurrence of Health Effects

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

The distribution of the total number of cancers is statistically predicted over the 100 years after 1 year of exposure, using data for the expected occurrence of cancer in the average population as a result of low-level radiation exposure. This curve is integrated over the operational lifetime of the facilities. After the shutdown of the facilities, except in the disposal stages, the releases do not continue and the level of radioactivity due to the releases will decrease dependant on their physical and environmental half-times. Estimates of the occurrence of severe hereditary effects during the next 12 generations were made using information presented in ICRP60.

3.4 Impacts of liquid releases of radionuclides

Depending on the site of the facility, liquid releases will occur into a river or the sea. The priority pathways for aquatic releases are the use of the water for drinking and irrigation, and the consumption of fish and other marine food products. The pathway is broadly similar to that shown in Figure AII.1 for atmospheric releases. For the freshwater environment exposure is possible through consumption of fish, and of crops irrigated by the water into which the liquid waste has been discharged. For the marine environment, the seafood and fish harvested for human consumption are the only priority pathway considered in this assessment. The other possible pathways involving the recreational use of the water and beaches do not contribute significantly to the population dose.

3.4.1 River

The dispersion of the releases in the river is typically modelled using a simple box model that assumes instantaneous mixing in each of the general sections of the river that have been defined. The upstream section becomes the source for the downstream section. River-specific characteristics, such as flow rate of water and sediments, transfer factors for water/sediments and water/fish, are needed for each section. The human use factors such as irrigation, water treatment and consumption, and fish consumption must also be taken into consideration.

The deposition of the radionuclides in the irrigation water to the surface of the soil and transfer to agricultural produce is assumed to be the same as for atmospheric deposition.
The ingestion pathway doses are calculated in the same way as described above for the atmospheric pathway. For aquatic releases, it is difficult to calculate independent local and

regional collective doses without creating extremely simplified and probably incorrect food distribution scenarios. Therefore, the local and regional collective doses are reported in the regional category. The estimation of health effects also follows the same methodology as described in the section above.

3.4.2 Sea

To evaluate the collective dose due to consumption of seafood and marine fish, a compartment model which divides the northern European waters into 34 sections was used for the original French implementation. This model takes into account volume interchanges between compartments, sedimentation, and the radionuclide transfer factors between the water, sediment, fish, molluscs, crustaceans, and algae, and the tons of fish, molluscs, crustaceans and algae harvested for consumption from each compartment. For the regional collective dose, it is assumed that the edible portion of the food harvested in the northern European waters is consumed by the European population before any surplus is exported globally. Due to the difficulty in making assumptions for the local consumption, the local collective dose is included in the regional results.

The risk estimates and monetary evaluation of this pathway uses the same methodology as the other pathways.

3.5 Impacts of releases of radionuclides from radioactive waste disposal sites

The land-based facilities designed for the disposal of radioactive waste, whether for low-level waste or high-level waste, are designed to provide multiple barriers of containment for a time period considered reasonable relative to the half-life of the waste. This environmental transfer pathway is again similar to that shown in Figure AII.1, though in this case emissions arise from leakage from the containers in which waste material is stored. It is assumed that with the normal evolution of the site with time, the main exposure pathway for the general public will be the use of contaminated ground water for drinking or irrigation of agricultural products.

The leakage rate and geologic transport of the waste must be modelled for the specific facility and the specific site. The global doses due to the total release of H-3, C-14 and I-129 are estimated assuming that ultimately the total inventory of wastes are released into the sub-surface environment. As is done for the other pathways, it is assumed that the local population and their habits remain the same for the 100,000-year time period under consideration for the disposal sites. This time limit takes into account disposal of all the radionuclides except long-lived I-129.

3.6 Impacts of accidental atmospheric releases of radionuclides

The methodology used to evaluate impacts due to accidental releases is 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:

$$\mathbf{Risk} = \sum \mathbf{P_i} \, . \, \mathbf{C_i}$$

3.6.1 Transportation accidents

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

The atmospheric dispersion of the release is modelled using a Gaussian plume puff model. The toxicological effects of the releases (specifically UF_6) are estimated using the LD_{50} (lethal dose for 50% of the exposed population) to estimate the number of expected deaths and a dose-response function for injuries due to the chemical exposure. The radiological impacts are estimated with the same methodology described for the atmospheric release pathway. The expected number of non-radiological impacts, such as death and physical injury due to the impact of the accident, are also included.

3.6.2 Severe Reactor Accidents

The public health impacts and economic consequences of the releases can be estimated using available software such as COSYMA (Ehrhardt and Jones, 1991), which was produced for the EC. The impact pathway must be altered to take account of the introduction of countermeasures for the protection of the public (decontamination, evacuation, food restrictions, changes in agricultural practices, etc.). The economic damages from the implementation of the countermeasures and the agricultural losses are calculated by COSYMA using estimates of the market costs.

It has to be noted that the use of this type of methodology does not necessarily include all the social costs that would result after a severe accident. Further work is required on this subject.

3.7 Occupational impacts from exposure to radiation

The legislation governing protection of workers from radiation requires direct monitoring and reporting of the doses received by the workers. The availability of such data means that it is not necessary to model exposure. The dose-response relationships are based on international recommendations of ICRP 60. The factors, or dose response functions, used to predict the expected occurrence of cancer over a lifetime or severe hereditary effects in future generations per unit exposure received by the workers are 0.04 fatal cancers per manSv and 0.006 severe hereditary effects in future generations per manSv.

The fraction of cancers that would be expected to be non-fatal are calculated based on the expected number of fatal cancers and the lethality fractions in the worker population reported for 9 categories of cancer reported in ICRP60. The different age and sex distributions found in the working population compared to the general public slightly changes the expected occurrence of disease.

3.8 Impacts of transportation on human health

The priority impact pathway from accident-free transportation operations in the nuclear fuel cycle is external exposure from the vehicle containing the radioactive material. Models such as the International Atomic Energy Agency's INTERTRAN code are available that take into account the content of the material transported, the type of container, mode of transport (road or rail), the distance travelled, and the number of vehicle stops at public rest stations along the highway (for road transportation).

3.9 Monetary valuation

Valuation data have been provided by Markandya (European Commission, 1998) (Table AII.?).

Table AII.? Valuation data relevant to assessment of radiological health impacts.

ENDPOINT	VALUE (ECU, 1995)	ESTIMATION METHOD AND COMMENTS
Fatalities (per case)	3,100,000	Average of estimates of the value of statisitcal life
Chronic Illness (VSC)	1,200,000	CVM in US estimating WTP. Inflation adjustment made
Non fatal Cancer	450,000	US study revised for inflation.
Malignant Neoplasms	450,000	Suggested valuing as non-fatal cancer.

4. Carcinogenic Effects of Dioxins and Trace Metals

The basic impact assessment approach used in ExternE for macropollutants (see above) is still valid for the micropollutants - after all it simply seeks to quantify the pathway from emission to impact and monetary damage. However, the step in which incremental exposure of the stock at risk is quantified requires elaboration to account for both direct and indirect exposure. The range of possible exposure pathways is shown in Table 1.

Table 1. Exposure pathways for persistent micropollutants.

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

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

4.1 Dioxins and Dibenzofurans

The dioxins are a family of 75 chlorinated tricyclic aromatic compounds, to which are often added 125 closely related compounds, the polychlorinated dibenzofurans. Several of these are highly toxic and they may also be carcinogenic. Their toxicity is illustrated by concern in spite of their emission levels being of the order of pg (10^{-12} g) per Nm^3, contrasted with levels greater than µg (10^{-6}) per Nm^3 for the other air pollutants of interest. For our purposes, analysis can be simplified using internationally accepted toxic equivalence factors (TEFs) relating the toxicity of other dioxins to 2,3,7,8 - tetrachlorodibenzodioxin (TCDD) (which is believed to be the most toxic dioxin) (NATO/CCMS, 1988). The aggregate figure of dioxin emissions, referred to as the toxic equivalence quotient (I-TEQ), is calculated by summing the products of mass of emission and TEF for each species.

4.1.1 Threshold levels

There is considerable debate about thresholds for the effects of dioxins on human health. Of particular note is the apparent divergence in opinion between Europe, where thresholds for carcinogenic and non-carcinogenic impacts of dioxins are generally accepted, and the USA, where no (or extremely low) threshold is assumed. Positions on both sides of the Atlantic are under review. Recent reviews for governments in France, the UK, and Germany all concluded that a threshold exists.

The position of the World Health Organisation (French Academy of Sciences, 1995) is that the tolerable daily intake (TDI) is 10 $pg/kg_{bw}\cdot day$ (10^{-12}g per kg body weight per day). The TDI represents an average lifetime dose, below which damage is considered unlikely. Calculation of the TDI involves the use of safety factors, which is illustrated in Table 3. Safety factors reflect the uncertainty involved in extrapolating data between species and also the perceived severity of the effect.

Table 3. Use of safety factors in setting guideline intake levels (DoE, 1989).

Effect	**NOEL[(a)] $pg/kg_{bw}\cdot day$**	**Safety factor**	**Guideline level $pg/kg_{bw}\cdot day$**
Immunotoxic	6000	100	60
Reprotoxic	120	100	1
Carcinogenic	10000	1000	10

[(a)] No observed effect level - derived from experimental data on sensitive animal species.

To calculate a lower estimate for dioxin damages we take the TDI of 10 $pg/kg_{bw}\cdot day$ as threshold. This is considered applicable to carcinogenic as well as non-carcinogenic effects, because dioxins are believed to be receptor-mediated carcinogens.

In contrast the position adopted by the US Environmental Protection Agency is for an acceptable daily intake about 1000 times lower based on an upper bound risk assessment of the level that carries a lifetime cancer risk of one in a million. The assumption that there is no threshold can thus be adopted for estimation of an upper estimate for damages, though it is emphasised that most expert opinion in Europe would follow the assumption that a threshold exists (although there is dispute as to the magnitude of that threshold).

4.1.2 Pathway analysis for dioxins

Considerable debate has surrounded the calculation of human exposure to dioxins from incineration. Whilst early studies concentrated on the direct (inhalation) exposure route, more recent analyses have modelled the transfer of the contaminants from the incinerator to the exposed population via most, or all of the routes shown in Table 1.

HMIP (1996) assessed the health risk from dioxins emitted to air by hypothetical municipal waste incineration plants located in rural and urban areas of the UK. The principal scenarios were based upon a plant size of 250,000 tonnes/year, with a dioxin emission concentration of 1.0 ng I-TEQ/Nm^3, but the analysis was extended to plant ranging from 100,000 to 500,000 tonnes/year, with dioxin emissions from 0.1 to 10 ng I-TEQ/Nm^3. Municipal waste incinerators meeting the current EU Directive will mostly emit within this range, though some go further, and some may have been exempted so far from the legislation. The study considered in detail the transfer of dioxins from air concentrations, via the soil, vegetation and animal food products, and via inhalation, to the human population in the vicinity of the plant. The dose received was calculated, across all plant sizes and emission concentrations, for average cases and a 'Hypothetical Maximally Exposed Individual' (HMEI). The HMEI is assumed to be located at the point of maximum ground level air dioxin concentration, consuming food which has been grown or reared at this location, drinking water from a reservoir also sited at this location, and exposed to such conditions over their entire lifetime. The HMEI therefore provides an ultra-conservative estimate of the risks faced by an individual.

The analysis covered background exposure and incremental exposure due to the incinerator. This allowed assessment of the relative importance of the different sources of the total dose, and, since the study used the WHO threshold value to assess health effects, an assessment of the net risk to the population from dioxin intake. Table 3 summarises the results for the plant emitting the highest levels of dioxins considered by HMIP.

Table 3. Summary of mean dioxin intakes for an adult HMEI* living close to an incinerator sited in urban and rural locations.

Exposure	**Urban Site pg I-TEQ kg.bw^{-1} day^{-1}**	**Rural Site pg I-TEQ kg.bw^{-1} day^{-1}**
Background	0.96	0.96
Incremental	0.73	0.12
Total	1.69	1.08

* Plant scenario: 500,000 t/y^{-1}, 10 ng I-TEQ Nm^{-3} emission concentration

It can be seen from Table 3 that even in the worst case considered, of an urban HMEI living near the largest plant emission considered by the study, the total intake does not approach the WHO threshold level. However, recent studies have suggested that the dioxin intake of an average breast-fed baby could be as high as 110 pg/kg/day at two months, falling to 25 pg/kg/day at ten months. Results from the HMIP study are not directly comparable (being averaged over a longer period), but also suggest exposure above the WHO recommended TDI of 10 pg/kg/day. However, we adopt the position of recent reviews (DoH, 1995), that when averaged over a lifetime, the cumulative effect of increased dioxin intake during breast feeding is not significant. The HMIP study concluded that emissions of dioxins from municipal waste incinerators operating to EU legislative standards do not pose a health risk to individuals, irrespective of the location and size of the incinerator or the exposed population.

Since the highest emissions limits used in the above study correspond to or exceed emissions from any incinerator likely to be built within the EU, it follows that no greater health effect should be seen from plant that meet existing Directives, assuming the threshold assumption made here is correct. Therefore estimated dioxin related damages would be zero (accepting that the present analysis is necessarily performed at too coarse a scale to pick up any individuals who, for whatever reason, have a far higher exposure to dioxin than the rest of the population).

There are two difficulties here. It is possible that breast-fed infants could be particularly sensitive to dioxins because of their developmental status. It is also possible that the threshold assumption adopted here is wrong, and that there is either no threshold, or that any threshold that does exist is so low as not to make a difference (in other words it is below typical exposure levels). In view of the genuine scientific uncertainty that exists, in particular the different attitudes between informed opinion in Europe and the USA, we therefore consider it appropriate to also consider the magnitude of the effect under the alternative assumption that there is no threshold (this would cover the full range of outcomes). Our view is that this is unlikely, but that the possibility cannot be excluded given the peculiar nature of dioxins (being present at minute levels, but having a very high toxicity). In this case it is not appropriate to restrict the analysis to the area in the vicinity of an incinerator, or to most exposed individuals. Everyone at risk of exposure from the specified plant should be considered. In practice this means consideration is given to people exposed to miniscule incremental levels of pollution. The probability of any individual being affected will be small. However, the aggregated damage, summed across the exposed population may well be significant.

For this sensitivity analysis we do not, however, consider it appropriate to carry out a full detailed assessment of all intake pathways, following the same level of detail as the HMIP study. This would be complicated by the necessary range of the assessment. Instead it is possible to simplify the analysis by calculating direct intake and multiplying this by an appropriate factor to obtain the total incremental dioxin dose. It is acknowledged that the uncertainty associated with this approach is significant. This uncertainty is reflected by the fact that the direct intake pathway provides only a small percentage of the total intake. The review by the US EPA (1994) cites a figure of 2% of the total dose arising through inhalation. Other published estimates are of a similar magnitude. This figure is assumed here to be the best available estimate. Total incremental exposure is thus calculated by multiplying inhaled dose by 50.

The inhaled dose is calculated from the ground level concentration by the following formula:

$$I = \frac{C \times IR \times ET \times EF \times ED}{BW \times AT}$$

Where

C: concentration (mg/m^3)
IR: inhalation rate (m^3/hour)
ET: exposure time (hours/day)
ED: exposure duration (years)
BW: body weight (kg)
AT: averaging time

EF: exposure frequency (hours/year)

A continuous exposure over 70 years is assumed. The factor EF * ED/AT is therefore unity, and equation (1) becomes:

$$\mathrm{I} = \frac{\mathrm{C} \times (\mathrm{IR} \times \mathrm{ET})}{\mathrm{BW}}$$

From this equation, and the expression of the I-TEQ per unit body weight it is apparent that body weight needs to be accounted for. Assumed values for body weight and IR*ET are shown in Table 4.

Table 4. Assumptions for calculating inhaled dose.

	Man	**Woman**	**Child**
Body weight (kg)	70	60	20
Inhalation volume (IR*ET) (m^3/day)	23	21	15

Assuming a 46.5%, 46.5% and 7% fraction of men, women and children respectively within the total population, it is possible to calculate a gender/age-weighted ‘Inhalation Factor’ IF;

$$IF = \frac{23m^3}{70kg \cdot d} \cdot 0.465 + \frac{21m^3}{60kg \cdot d} \cdot 0.465 + \frac{15m^3}{20kg \cdot d \cdot} \cdot 0.07 = 0.368m^3 / (kg \cdot d)$$

The relation between dose and concentration then is

$$I = C \cdot IF = C \cdot 0.368m^3 / (kg \cdot d)$$

However, the dose described by the above equation is the inhalation dose only. To estimate the total dose, we can use the estimates on the fraction of inhalation contributing to the total dose, as given in the IEH report. Thus, the total dose is estimated to be

$$I_{Total} = C \cdot \frac{IF}{InhalationFraction}$$

with e.g. an InhalationFraction of 0.02 for Dioxins (relative exposure via inhalation = 2%).

No-threshold assumption:
Unit risk factor from
LAI 1.4 per µg/m^3
leading to the following ERF implemented in EcoSense:
(1) No. of additional cancers = Δ Concentration [µg/m^3] * 1.4 * Population /70

Threshold assumption:

WHO ‘tolerable daily intake’: 10 pg/(kg_{BW}·d)

Using an InhalationFraction of 0.02 (relative exposure via inhalation = 2 %), the air concentration equivalent to the threshold dose is 5.4 E-7 µg/m^3.

Background:

UK (HMIP, 1996)	0.96 pg/(kg*d)	==>	5.22 E-8 µg/m^3
France (Rabl, 1996):			2.4 E-8 µg/m^3
Germany (LAI):	0.41 pg/(kg*d)	==>	2.2 E-8 µg/m^3

4.2 Impact Assessment for Heavy Metals

As is the case for dioxins, the heavy metals expelled from incinerators are persistent in the environment. In some cases direct and indirect exposure pathways would need to be considered. However, there is a constraint of the availability of exposure-response data that precludes assessment of any non-carcinogenic effect for most heavy metals.

Direct intake rates are calculated from ground level air concentration using the same approach as that adopted for dioxins (see above).

For those metals with a non-carcinogenic effect, the possibility of a health impact is assessed through comparison of total dose (background plus incremental) and the threshold value below which no effects will be seen. Due to the lack of dose-response data further quantification is not possible with 2 exceptions, for lead and mercury (though see notes below).

The specific approach applied to each of the heavy metals of most concern is described below. In most cases a selection of exposure-response functions are available, we suggest alternatives for sensitivity analysis. Assessments conducted so far have suggested that the effects of heavy metal emissions will be negligible, avoiding the need to identify any single function as the best available. Other heavy metals not listed here are regarded as less toxic and hence unlikely to produce effects larger than those for the elements listed here.

The general form of the exposure-response function is as follows for all cancer effects;
No. of additional cancers = Δ Concentration [µg/m^3] * unit risk factor * Population /70

The factor of 70 annualises lifetime risk (assuming an average longevity of 70 years).

4.2.1 Cadmium

Cancer

Unit risk factors from

ATSDR (1989)	0.0018 per µg/m^3
LAI	0.012 per µg/m^3

Non-carcinogenic effects

Threshold:

WHO-Guidelines (1987):

Rural areas: present levels of < 1-5 ng/m^3 should not be allowed to increase

Urban areas: levels of 10-20 ng/m^3 may be tolerated.

Background:

According to WHO (1987); '*Cadmium concentrations in rural areas of Europe are typically a few ng/m^3 (below 5 ng/m^3); urban values range between 5 and 50 ng/m^3, but are mostly not higher than 20 ng/m^3.*'

No dose-response function is available for non-carcinogenic effects, so quantification has not been performed. However, it is noted that exceedance of the WHO guidelines does happen, so effects cannot be ruled out.

4.2.2 Mercury

Cancer
Generally not classified as carcinogenic.

Non-carcinogenic effects
Threshold:
From US-EPA: 0.3 µg/m^3

Background:
WHO-Air Quality Guidelines 1987:
rural areas: 2-4 ng/m^3
urban areas: 10 ng/m^3

Reported thresholds are so much higher than background air exposures that effects linked to air emissions from fuel cycle activities seem unlikely in all places apart from those with high mercury levels associated with certain industrial processes (which may or may not be linked to the energy sector), or high historical contamination.

4.2.3 Arsenic

Cancer
Unit risk factors from

WHO (1987)	0.003 per µg/m^3
US-EPA (1996)	0.0002 per µg/m^3
LAI	0.004 per µg/m^3

Non-carcinogenic effects
Threshold:
US-EPA 0.3 µg/(kg$_{BW}$·d)

Using the equations derived above and an InhalationFraction of 0.004 (relative exposure via inhalation = 0.4 %), the air concentration equivalent to the threshold dose is 3.3 ng/m^3

Background:
WHO-Air Quality Guidelines 1987:
rural areas: 1-10 ng/m^3
urban areas: < 1 µg/m^3
France (see Rabl, 1996): 1 - 4 ng/m^3
LAI (Germany)

rural areas: < 5 ng/m^3
urban areas: < 20 ng/m^3

Thus it is possible that background levels might exceed threshold, but there are no exposure response functions available for impact quantification.

4.2.4 Chromium

Cancer
Unit risk factor from
WHO (1987) 0.04 per µg/m^3

Non-carcinogenic effects
Not analysed: acute toxic effects typically only occur at high levels that are typically only encountered occupationally.

4.2.5 Nickel

Cancer
Unit risk factors from
WHO (1987) 0.0004 per µg/m^3
US-EPA (1996) 0.004 per µg/m^3
Non-carcinogenic effects
Threshold:
ATSDR (1996) 0.02 mg/(kg$_{BW}$·d)

Using the equations derived above, and an InhalationFraction of 0.003 (relative exposure via inhalation = 0.3 %), the air concentration equivalent to the threshold dose is 0.16 µg/m^3 or 160 ng/m^3.

Background:
WHO (1987)
rural areas: 0.1 - 0.7 ng/m^3
urban areas: 3 - 100 ng/m^3
industrial areas: 8 - 200 ng/m^3

Again there is the possibility that some individuals will be exposed to levels above the threshold, though as before, in the absence of a dose-response function a quantification of damages is not possible.

5. Occupational Health Issues (Disease and Accidents)

5.1 Sources of data

Results for this category of effects are calculated from data (normalised per unit of fuel output, or fuel chain input) on the incidence of disease and accidents in occupations linked to each fuel chain. Given advances in health and safety legislation in many countries it is essentail that the data used are, so far as possible;

- recent
- representative
- specific to the industry concerned
- specific to the country concerned

It can sometimes be difficult to ensure that data are ‘representative’. Fatal work related accidents are fortunately much rarer in many countries nowadays than they used to be. It is thus usually necessary to use data for a number of years aggregated at the national level (rather than at the level of a single site, which is the basis for most of our analysis) to obtain a robust estimate of the risk of a fatal accident. This will inevitably increase risk estimates for some sites where fatal accidents have never been recorded. The fact that such an accident has never ocurred at a particular site does not mean that the risk of a fatal accident is zero. Conversely, risks could be seriously exaggerated if undue weight were given to severe events which tend to happen very infrequently, such as the Piper Alpha disaster in the North Sea. By averaging across years and sites (up to the national level) such potential biases are reduced.

It is possible that analysis will be biased artificialy against some fuel chains (particularly coal and nuclear). This problem arises because the occupational effects in for example the oil and gas industry may not have been studied sufficiently long enough to identify real problems (remembering that the North Sea oil industry is little more than 25 years old). Another problem in looking at long term effects on workers relates to their mobility in some industries. It is beyond the scope of this study to correct any such bias, but we flag up the potential that it might exist.

The best sources of data are typically national health and safery agencies, and bodies such as the International Labor Organisation.

Given that everyone is exposed to risk no matter what they do, there is an argument for quantifying risk nett of an average for the working population as a whole. For the most part this has been found to make little difference to the analysis, with the exception of analysis of the photovoltaic fuel cycle in Germany (Krewitt *et al*, 1995). However, it does introduce a correction for certain labour intensive activities of low risk.

There has been some misunderstanding about the assessment of occupational health effects under the ExternE Project. The first series of reports published under the project reported (for the most part) only effects occurring in the country where the power station was based. The reason for doing this was that fuel chain operations were largely confined to the countries in which the power plants were located for the cases under investigation at the time the reports were written (coal in UK and Germany; nuclear in France; gas in the UK, lignite in Germany, hydro in Norway). It was not intended to imply that occupational health effects outside the country in which the power station was located could or should be ignored. In fact to do so would introduce an artificial bias against, in particular, indigenous extraction of fuels.

Within the present phase of the study a broader assessment of damages has been necessary, partly because of changing market conditions (the UK for example is no longer entirely dependent on domestic coal mines), and partly because of the inclusion of cases where the countries concerned do not have an indigenous supply of fuel. Occupational health data have therefore been collected for as many countries and fuel chains as possible in the present phase of the study.

5.2 Valuation

The following data have been provided by Markandya (European Commission, 1998).

ENDPOINT	VALUE (ECU, 1995)	ESTIMATION METHOD AND COMMENTS
Fatalities	3,100,000	Average of estimates of the value of statisitcal life
Chronic Illness (VSC)	1,200,000	CVM in US estimating WTP. Inflation adjustment made
Non fatal Cancer	450,000	US study revised for inflation.
Malginant Neoplasms	450,000	Suggested valuing as non-fatal cancer.
Occupational Injuries (minor)	78	French compensation payments, increased for inflation.
Occupational Injuries (major)	22,600	French compensation payments increased for inflation.
Workers & Public Accidents (minor)	6,970	TRL (1995). New estimates.
Workers & Public Accidents (major)	95,000	TRL (1995). New estimates.

Valuation of damages in non-EU Member States is carried out adjusting the valuation data using PPP (purchasing power parity) adjusted GDP (European Commission, 1998). Such adjustment is much less controversial in the context of occupational health effects than for (e.g.) global warming damage assessment, because the decision to increase exposure to occupational risk is taken within the country whose citizens will face the change in risk.

A particular problem for assessment of occupational damages relates to the extent that these damages might be internalised, for example through insurance and compensation payments, higher wage rates, etc. In part, internalisation requires workers to be fully mobile (so that they have a choice of occupation) and fully informed about the risks that they face. Available evidence suggests that internalisation is rarely, if ever, complete. With a lack of data on the extent to which internalisation is achieved, we report total damages instead.

6. Accidents Affecting Members of the Public

Most accidents affecting members of the public seem likely to arise from the transport phase of fuel chains, and from major accidents (for discussion of which see European Commission, 1998). The same issues apply to assessment of accidents concerning the general public as for occupational accidents; data must be representative, recent, and relevant to the system under investigation.

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APPENDIX III: AIR POLLUTION EFFECTS ON MATERIALS

1. Introduction

The effects of atmospheric pollutants on buildings provide some of the clearest examples of damage related to the combustion of fossil fuels. Pollution related damage to buildings includes discoloration, failure of protective coatings, loss of detail in carvings and structural failure. In the public arena most concern about pollutant damage to materials has focused on historic monuments. However, impacts of air pollution on materials are, of course, not restricted to buildings of cultural value. They have also been recorded on modern buildings and to other types of materials such as textiles, leather and paper. Given the relative abundance of modern buildings compared to older ones, it may be anticipated that damages to the former will outweigh those to the latter. However, without data on the way that people value historic monuments the relative importance of damage to the two types of structure is a matter of speculation.

This Appendix reviews the methodology used in the assessment of material damages within the ExternE Project. The analysis presented here is limited to the effects of acidic deposition on corrosion because of a lack of data on other damage mechanisms. As elsewhere in these Appendices, we attempt here only to provide an overview of the methodology used and the sources of data. Further details are given in the updated ExternE Methodology report (European Commission, 1998).

2. Stock at Risk Data

The stock at risk is derived from data on building numbers and construction materials taken from building survey information. Such studies are generally performed for individual cities; these can then be extrapolated to provide inventories at the national level. For countries data are not available, values must be extrapolated from elsewhere although this inevitably results in lower accuracy. The EcoSense model contains data from a number of such surveys that have been conducted around Europe. Where possible country-specific data has been used. Unfortunately this is not available for all countries, so some extrapolation is necessary. For the most part it is assumed that the distribution of building materials follows the distribution of population. Sources of data are as follows;

Eastern Europe:

Kucera *et al* (1993b), Tolstoy *et al* (1990) - data for Prague

Scandinavia:

Kucera *et al*, 1993b; Tolstoy *et al*, 1990 - data for Stockholm and Sarpsborg

UK, Ireland:

Ecotec (1996), except galvanised steel data, taken from European Commission (1995); data for UK extrapolated to Ireland

Greece:

NTUA (1997)

Germany, other Western Europe:

Hoos *et al* (1987) - data for Dortmund and Köln

3. Meteorological, Atmospheric and Background Pollution data

The exposure-response functions require data on meteorological conditions. Of these, the most important are precipitation and humidity. The following sources of data have been used;

For the UK:

UKMO (1977) - precipitation; UKMO (1970) - relative humidity; UKMO (1975) - estimated percentage of time that humidity exceeds critical levels of 80%, 85% and 90%; Kucera (1994) - UK background ozone levels.

For Germany;

Cappel and Kalb (1976), Kalb and Schmidt (1977), Schäfer (1982), Bätjer and Heinemann (1983), Höschele and Kalb (1988) - estimated percentage of time that relative humidity exceeds 85%; Kucera (1994) - other data.

For other countries data were taken from Kucera (1994).

4. Identification of Dose-Response Functions

Exposure response functions for this project come from 3 main studies; Lipfert (1987; 1989), the UK National Materials Exposure Programme (Butlin *et al*, 1992a; 1992b; 1993), and the ICP UN ECE Programme (Kucera, 1993a, 1993b, 1994), a comparison of which is shown in Table 1.

Table 1. Comparison of the Dose-Response Functions for Material Damage Assessment.

	Kucera	**Butlin**	**Lipfert**
Exposure time	4 years	2 years	-
Experimental technique	Uniform	Uniform	Meta analysis
Region of measurement	Europe	UK	-
Derivation of relationships	Stepwise linear regression	Linear regression	Theoretical

This section describes background information on each material and list the dose-response functions we have considered. The following key applies to all equations given:

ER	=	erosion rate (µm/year)
P	=	precipitation rate (m/year)
SO_2	=	sulphur dioxide concentration (µg/m^3)
O_3	=	ozone concentration (µg/m^3)
H^+	=	acidity (meq/m^2/year)
R_H	=	average relative humidity, %
f_1	=	$1-\exp[-0.121.R_H/(100-R_H)]$
f_2	=	fraction of time relative humidity exceeds 85%
f_3	=	fraction of time relative humidity exceeds 80%
TOW	=	fraction of time relative humidity exceeds 80% and temperature >0°C
ML	=	mass loss (g/m^2) after 4 years

MI = mass increase (g/m^2) after 4 years
CD = spread of damage from cut after 4 years, mm/year
Cl^- = chloride deposition rate in $mg/m^2/day$
$Cl_{(p)}^-$ = chloride concentration in precipitation (mg/l)
D = dust concentration in $mg/m^2/day$

In all the ICP functions, the original H^+ concentration term (in mg/l) has been replaced by an acidity term using the conversion:

$$P \cdot H^+ \text{ (mg/l)} = 0.001 \cdot H^+ \text{ (acidity in meq/m}^2\text{/year)}$$

To convert mass loss for stone and zinc into an erosion rate in terms of material thickness, we have assumed respective densities of 2.0 and 7.14 tonnes/m^3.

4.1 Natural stone

The ability of air pollution to damage natural stone is well known, and hence will not be debated further in this report. A number of functions are available;

Lipfert - natural stone: $ER = 18.8 \cdot P + 0.052 \cdot SO_2 + 0.016 \cdot H^+$ [1]

Butlin - Portland limestone: $ER = 2.56 + 5.1 \cdot P + 0.32 \cdot SO_2 + 0.083 \cdot H^+$ [2]

ICP - unsheltered limestone (4 years):

$ML = 8.6 + 1.49 \cdot TOW \cdot SO_2 + 0.097 \cdot H^+$ [3]

Butlin - sandstone: $ER = 11.8 + 1.3 \cdot P + 0.54 \cdot SO_2 + 0.13 \cdot H^+ - 0.29 \cdot NO_2$ [4]

ICP - unsheltered sandstone (4 years):

$ML = 7.3 + 1.56 \cdot TOW \cdot SO_2 + 0.12 \cdot H^+$ [5]

ICP - sheltered limestone (4 years):

$MI = 0.59 + 0.20 \cdot TOW \cdot SO_2$ [6]

ICP - sheltered sandstone (4 years):

$MI = 0.71 + 0.22 \cdot TOW \cdot SO_2$ [7]

Our assessment has relied on functions [3] and [5], because of the duration of reported exposure, and the fact that the work led by Kucera has been conducted across Europe.

4.2 Brickwork, mortar and rendering

Observation in major cities suggests that brick is unaffected by sulphur dioxide attack. However, although brick itself is relatively inert to acid damage, the mortar component of brickwork is not. The primary mechanism of mortar erosion is acid attack on the calcareous cement binder (UKBERG, 1990; Lipfert, 1987). Assuming that the inert silica aggregate is lost when the binder is attacked, the erosion rate is determined by the erosion of cement. Functions are approximated from those derived for sandstone [4] and [5], as specific analysis has not been carried out on mortar.

4.3 Concrete

The major binding agent in most concrete is an alkaline cement which is susceptible to acid attack. Potential impacts to concrete include soiling/discoloration, surface erosion, spalling and enhanced corrosion of embedded steel. However, for all these impacts (with the exception of surface erosion) damages are more likely to occur as a result of natural carbonation and ingress of chloride ions, rather than interaction with pollutants such as SO_2. Effects on steel embedded in reinforced concrete are possible, but no quantitative information exists for these processes. In view of this damage to concrete has not been considered in the study.

4.4 Paint and polymeric materials

Damages to paint and polymeric materials can occur from acidic deposition and from photochemical oxidants, particularly ozone. Potential impacts include loss of gloss and soiling, erosion of polymer surfaces, loss of paint adhesion from a variety of substrates, interaction with sensitive pigments and fillers such as calcium carbonate, and contamination of substrate prior to painting leading to premature failure and mechanical property deterioration such as embrittlement and cracking particularly of elastomeric materials.

The most extensive review in this area is from the USA (Haynie, 1986). This identifies a 10-fold difference in acid resistance between carbonate and silicate based paints. The dose-response functions are as follows, in which t_c = the critical thickness loss, about 20 µm for a typical application:

Haynie - carbonate paint:

$$\Delta ER/t_c = 0.01 \cdot P \cdot 8.7 \cdot (10^{-pH} - 10^{-5.2}) + 0.006 \cdot SO_2 \cdot f_1 \qquad [8]$$

Haynie - silicate paint:

$$\Delta ER/t_c = 0.01 \cdot P \cdot 1.35 \cdot (10^{-pH} - 10^{-5.2}) + 0.00097 \cdot SO_2 \cdot f_1 \qquad [9]$$

There are problems with the application of these functions. These are discussed in more detail by European Commission (1998). However, in the absence of superior data the function on carbonate paint has been applied.

4.5 Metals

Atmospheric corrosion of metals is well accepted. Of the atmospheric pollutants, SO_2 causes most damage, though in coastal regions chlorides also play a significant role. The role of NO_x and ozone in the corrosion of metals is uncertain, though recent evidence (Kucera, 1994) shows that ozone may be important in accelerating some reactions.

Although dose-response functions exist for many metals, this analysis is confined to those for which good inventory data exists; steel, galvanised steel/zinc and aluminium. Other metals could be important if the material inventories used were more extensive, quantifying for example copper used in historic monuments. Steel is virtually always used coated with paint when not galvanised. The stock of steel in our inventories has therefore been transferred to the paint stock at risk.

4.5.1 Zinc and galvanised steel

Zinc is not an important construction material itself, but is extensively used as a coating for steel, giving galvanised steel. Zinc has a lower corrosion rate than steel, but is corroded in preference to steel, thereby acting as a protective coating. Despite a large number of studies of zinc corrosion over many years, there still remains some uncertainty about the form of the dose-response function. One review (UKBERG, 1990) identifies 10 different functions that assume time linearity, consistent with the expectation that the products of corrosion are soluble and therefore non-protective. However, other reviews (Harter, 1986 and NAPAP, 1990) identify a mixture of linear and non-linear functions. It is thus clear that uncertainties remain in spite of an apparent wealth of data. Further uncertainty arises from the recent introduction of more corrosion resistant zinc coatings onto the market. For this study, we have used the following functions, with particular emphasis on those reported by Kucera *et al* (1994) from the UNECE ICP (equations [12] and [13]).

Lipfert - unsheltered zinc (annual loss):

$$ML = [t^{0.78} + 0.46\log_e(H^+)]\cdot[4.24+0.55\cdot f_2\cdot SO_2+0.029\cdot Cl^-+0.029\cdot H^+] \qquad [10]$$

Butlin - unsheltered zinc (one year):

$$ER = 1.38 + 0.038\cdot SO_2\cdot + 0.48P \qquad [11]$$

ICP - unsheltered zinc (4 years):

$$ML = 14.5 + 0.043\cdot TOW\cdot SO_2\cdot O_3 + 0.08\cdot H^+ \qquad [12]$$

ICP - sheltered zinc (4 years):

$$ML = 5.5 + 0.013\cdot TOW\cdot SO_2\cdot O_3 \qquad [13]$$

To date, the assessments in the ExternE Project have not considered incremental ozone levels from fuel cycle emissions with respect to materials damage. These equations demonstrate that this may introduce additional uncertainty into our analysis.

4.5.2 Aluminium

Aluminium is the most corrosion resistant of the common building materials. In the atmosphere aluminium becomes covered with a thin, dense, oxide coating, which is highly protective down to a pH of 2.5. In areas where pollution levels are very high an average of equations [14] and [15] is recommended. Elsewhere simple corrosion of aluminium seems unlikely to be of concern. No functions are available for ‘pitting’ as a result of exposure to SO_2 which appears to be a more serious problem (Lipfert, 1987).

Lipfert - aluminium (annual loss):

$$ML = 0.2\cdot t^{0.99}\cdot(0.14\cdot f_3\cdot SO_2+0.093\cdot Cl^-+0.0045\cdot H^+-0.0013\cdot D)^{0.88} \qquad [14]$$

ICP - unsheltered aluminium (4 year):

$$ML = 0.85 + 0.0028\cdot TOW\cdot SO_2\cdot O_3 \qquad [15]$$

5. Calculation of repair frequency

We assume that maintenance is ideally carried out after a given thickness of material has been lost. This parameter is set to a level beyond which basic or routine repair schemes may be insufficient, and more expensive remedial action is needed. A summary of the critical thickness loss for maintenance and repair are shown in Table 2. The figures given in Table 2 represent averages out of necessity, though the loss of material will not be uniform over a building. Some areas of a building at the time of maintenance or repair would show significantly more material loss than indicated by the 'critical thickness', and others less. It may also be expected that the maintenance frequency would be dictated most by the areas that are worst damaged.

Table 2. Averages of country-specific critical thickness losses for maintenance or repair measures assumed in the analysis

Material	Critical thickness loss
Natural stone	4 mm
Rendering	4 mm
Mortar	4 mm
Zinc	50 µm
Galvanised steel	50 µm
Paint	50 µm

6. Estimation of economic damage (repair costs)

The valuation of impacts should ideally be made from the willingness to pay to avoid the incremental damage. No assessments of this type are available. Instead, repair/replacement costs of building components are used as a proxy estimate of economic damage. The main complication here relates to uncertainty about the time at which people would take action to repair or maintain their property. We assume that everyone reacts rationally, in line with the critical thickness losses described in Section 5. It is recognised that some people take action for reasons unrelated to material damage (e.g. they decide to paint their house a different colour). The effect of air pollution in such cases would be zero (assuming it has not caused an unpleasant change in the colour of the paint!). However, other people delay taking action to repair their buildings. If this leads to secondary damage mechanisms developing, such as wood rot following paint failure that has been advanced through exposure to air pollution, additional damage will arise. Given the conflicting biases that are present and a lack of data on human behaviour, the assumption followed here seems justified.

It is necessary to make some assumptions about the timing of the costs. For a building stock with a homogeneous age distribution, the incidence of repair and replacement costs will be uniform over time, irrespective of the pollution level. The repair/replacement frequency is then an adequate basis for valuation with costs assumed to occur in the year of the emission. The reference environment building stock corresponds relatively well to the requirement of a homogeneous age distribution. There are some exceptions, where the age distribution, and consequently replacement time distribution, are more strongly concentrated in some periods. However, the error in neglecting this effect will be small for analysis across Europe compared to other uncertainties in the analysis.

Estimates for the repair costs have been taken from different sources. For the UK estimated repair costs are taken from unit cost factors for each of the materials for which assessment was performed. These figures are based on data from ECOTEC (1986) and Lipfert (1987). For Germany repair costs have been obtained from inquiries with German manufacturers. Finally, damage costs given in a study for Stockholm, Prague and Sarpsborg (Kucera *et al*, 1993b) are also considered. Table 3 summarises the damage costs used in this analysis in 1995ECU.

Table 3 Repair and maintenance costs [ECU/m^2] applied in analysis

Material	ECU/^{m2}
Zinc	25
Galvanised steel	30
Natural stone	280
Rendering, mortar	30
Paint	13

Identical repair costs are used for all types of repainting, whether on wood surfaces, steel, galvanised steel, etc. This is likely to underestimate impacts, as some paints such as the zinc rich coatings applied to galvanised steel will be more expensive than the more commonly applied paints for which the cost data are strictly appropriate.

7. Estimation of soiling costs

Soiling of buildings results primarily from the deposition of particulates on external surfaces. Three major categories of potential damage cost may be identified; damage to the building fabric, cleaning costs and amenity costs. In addition, there may be effects on building asset values, as a capitalised value of these damages.

Cleaning costs and amenity costs need to be considered together. Data on the former is, of course, easier to identify. In an ideal market, the marginal cleaning costs should be equal to the marginal amenity benefits to the building owner or occupier. However, markets are not perfect and amenity benefits to the public as a whole lie outside this equation. It is therefore clear that cleaning costs will be lower than total damage costs resulting from the soiling of buildings. In the absence of willingness to pay data, cleaning cost are used here as an indicator of minimum damage costs.

Where possible a simple approach has been adopted for derivation of soiling costs. For example, in the analysis of UK plants, we assume that the total impact of building soiling will be experienced in the UK. The total UK building cleaning market is estimated to be £80 million annually (Newby *et al*, 1991). Most of this is in urban areas and it is assumed that it is entirely due to anthropogenic emissions. Moreover, it can reasonably be assumed that cleaning costs are a linear function of pollution levels, and therefore that the marginal cost of cleaning is equal to the average cost.

Different types of particulate emission have different soiling characteristics (Newby *et al*, 1991). The appropriate measure of pollution output is therefore black smoke, which includes this soiling weighting factor, rather than particulates, which does not. UK emissions of black smoke in 1990 were 453,000 tonnes (DOE, 1991). The implied average marginal cost to building cleaning is therefore around 300 ECU/tonne. This value is simply applied to the plant output. The method assumes that emission location is not important; in practice, emissions from a plant outside an urban area will have a lower probability of falling on a building. However, given the low magnitude of the impact, further refinement of the method for treatment of power station emissions was deemed unnecessary.

Results from the French implementation (European Commission, 1995) have shown that for particulate soiling, the total cost is the sum of repair cost and the amenity loss. The results show that, for a typical situation where the damage is repaired by cleaning, the amenity loss is equal to the cleaning cost (for zero discount rate); thus the total damage costs is twice the cleaning cost. Data from the same study shows cleaning costs for other European countries may be considerably higher than the UK values.

8. Uncertainties

Many uncertainties remain in the analysis. In particular, the total damage cost derived is sensitive to some parts of the analysis which are rather uncertain and require further examination. The following are identified as research priorities:

- Improvement of inventories, in particular; the inclusion of country specific data for all parts of Europe; disaggregation of the inventory for paint to describe the type of paint in use; disaggregation of the inventory for galvanised steel to reflect different uses; disaggregation of calcareous stone into sandstone, limestone, etc. In addition, alternatives to the use of population data for extrapolation of building inventories should be investigated.
- Further development of dose-response functions, particularly for paints, mortar, cement render, and of later, more severe damage mechanisms on stone;
- Assessment of exposure dynamics of surfaces of differing aspect (horizontal, sloping or vertical), and identification of the extent to which different materials can be considered to be sheltered;
- Definition of service lifetimes for stone, concrete and galvanised steel;
- Integration of better information on repair techniques;
- Data on cleaning costs across Europe;
- Improvement of awareness of human behaviour with respect to buildings maintenance;
- The extension of the methodology for O_3 effects, including development of dose-response functions and models atmospheric transport and chemistry.

Although this list of uncertainties is extensive, it would be wrong to conclude that our knowledge of air pollution effects on buildings is poor, certainly in comparison to our knowledge of effects on many other receptors. Indeed, we feel that the converse is true; it is because we know a great deal about damage to materials that we can specify the uncertainties in so much detail.

Some of these uncertainties will lead to an underestimation of impacts, and some to an overestimation. The factors affecting galvanised steel are of most concern given that damage to it comprises a high proportion of total materials damage. However, a number of potentially important areas were excluded from the analysis because no data were available. In general, inclusion of most of these effects would lead to greater estimates of impacts. They include:

- Effects on historic buildings and monuments with "non-utilitarian" benefits;
- Damage to utilitarian structures that were not included in the inventory;
- Damage to paint work through mechanisms other than acid erosion;
- Damage to reinforcing steel in concrete;
- Synergies between different pollutants;
- Impacts of emissions from within Europe on buildings outside Europe;
- Impacts from ozone;
- Macroeconomic effects.

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APPENDIX IV: ANALYSIS OF ECOLOGICAL IMPACTS

1. Introduction

Fuel chain activities are capable of affecting ecosystems in a variety of ways. This Appendix deals specifically with effects of air pollution on crop yield, on forest health and productivity, and effects of nitrogen on critical loads exceedence.

An approach for the analysis of acidification effects on freshwater fisheries was described earlier (European Commission, 1995a). This has not been implemented further because of a lack of data in many areas. However, work in this area is continuing, and it is anticipated that further progress will be made in the near future.

The are expected to be numerous effects of climate change, particularly concerning coastal regions and species range. These are partly dealt with in the assessment of global warming (Appendix V and European Commission, 1998a).

Approaches for dealing with local impacts on ecology, for example, effects of transmission lines on bird populations and were discussed in the earlier ExternE report on the hydro fuel cycle (European Commission, 1995b). Assessment of such effects is complicated by the extreme level of site specificity associated with the damage. In most cases in EU Member States local planning regulations should reduce such damage to a negligible level. However, inevitably there are sites where significant ecological resources are affected.

2. Air Pollution Effects on Crops

2.1 SO_2 Effects

A limited number of exposure-response functions dealing with direct effects of air pollution on crops are available. These were described in depth in the earlier report on the ExternE methodology (European Commission, 1995a). Baker *et al* (1986) produce the following function from work on winter barley;

$$\% \text{ Yield Loss} = 9.35 - 0.69(SO_2) \qquad (1)$$

Where SO_2 = annual mean SO_2 concentration, ppb.

One problem with the study by Baker *et al* and other work in the area is that experimental exposures rarely extend below an SO_2 concentration of about 15 ppb. This is assumed to correspond to a 0% yield reduction. It has been demonstrated in a large number of experiments that low levels of SO_2 are capable of stimulating growth, and so it cannot be assumed that there is no effect on yield below 15 ppb, nor can it be assumed that any effect will be detrimental. As few rural locations in Europe experience SO_2 levels greater than 15 ppb, equation (1) is not directly applicable. To resolve this, a curve was estimated that fitted the following criteria, producing an exposure-response of the form suggested by Fowler *et al* (1988):

1. 0% yield reduction at 0 ppb and also at the value predicted by equation (1);
2. Maximum yield increase at an SO_2 concentration midway between the 2 values for which 0% yield effect is predicted from (1);

3. The experimentally predicted line to form a tangent to this curve at the point corresponding to 0% yield change with SO_2 concentration > 0..

This approach gave the following set of exposure-response functions, in which the concentration of SO_2 is expressed in ppb and y = % yield loss;

Baker modified: $y = 0.74(SO_2) - 0.055(SO_2)^2$ (from 0 to 13.6 ppb) (2a)

$y = -0.69(SO_2) + 9.35$ (above 13.6 ppb) (2b)

An iluustration of the extrapolation procedure is shown in Figure 1.

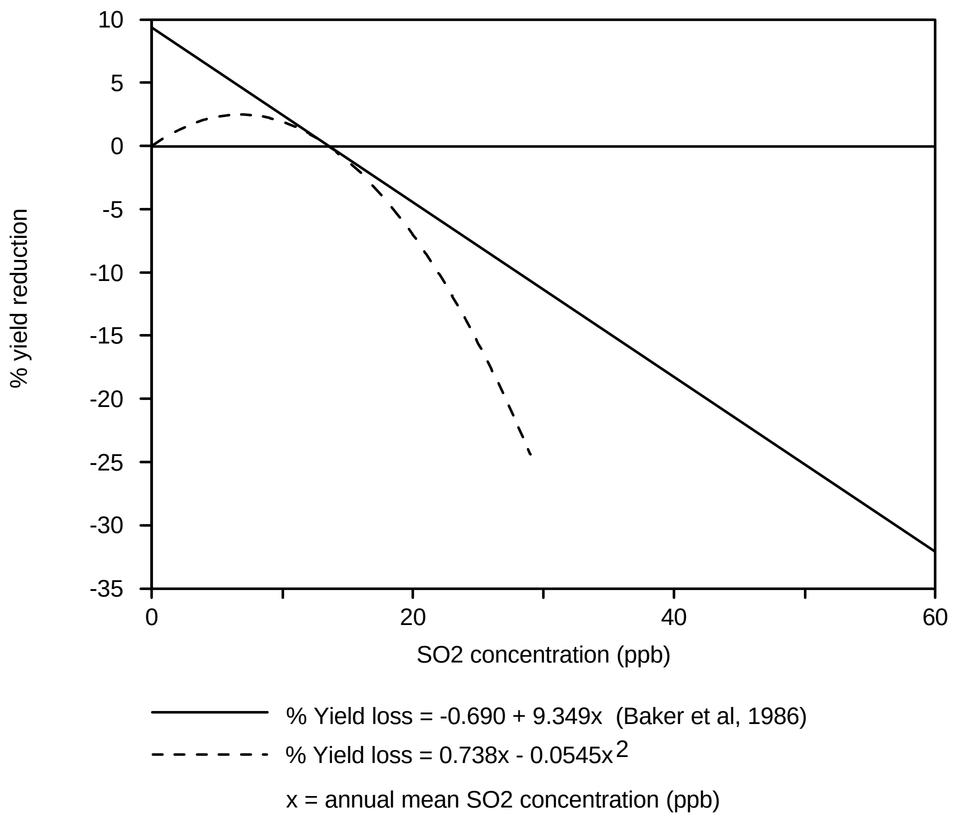

Figure 1. Extrapolation of exposure-response functions below the lowest exposure level used experimentally.

It is important to ascertain precisely what stresses may be included in any exposure-response relationship. Baker *et al* (1986) reported that weather conditions varied greatly between years in their experiment; *'1983/4 had an ordinarily cold winter and a dry, sunny summer, but the winter of 1984/5 was severe in January and February and the summer was dull and wet'.* However, there was a high degree of consistency in their results. Further details are as follows; Mean O_3 and NO_x concentrations were around 19 ppb and 24 ppb, respectivel; The soil was a sandy loam; Management practises in this work reflected those typical of local farms, fertiliser and agrochemicals being applied at the same times and rates. No records of pest or pathogen performance are given in the paper.

Weigel *et al* (1990) studied several crop cultivars common in Germany. Two spring barley cultivars ('Arena', 'Hockey'), two bean cultivars ('Rintintin', 'Rosisty') and one rape cultivar ('Callypso') were exposed to five different SO_2 levels between 7 and 202 µg m^{-3} (2.5 - 70 ppb) in open-top chambers. Exposure periods ranged from 49 to 96 days. 8 h/daily mean O_3-concentration ranged between 14 and 19 µg m^{-3}. Daily means of NO_2 and NO concentrations were generally lower than 10 µg m^{-3}. Yield increases appeared in all SO_2 treatments for the rape cultivar compared with controls whereas beans and barley were quite SO_2 sensitive. The probable cause of the positive response of rape was the high sulphur demand of this species. Data for barley and beans were taken from this paper and used to calculate the following relationships (SO_2 in µg m^{-3}):

$$y = 4.92 - 0.26(SO_2) \quad (3)$$

$r^2 = 0.73$, $p < 0.001$, 20 data points for barley and beans
($y = 4.92 - 0.74(SO_2)$, SO_2 in ppb)

$$y = 10,92 - 0,31(SO_2) \quad (4)$$

$r^2 = 0.73$, $p < 0.01$, 10 data points for barley only
($y = 10.92 - 0.89(SO_2)$, SO_2 in ppb)

$$y = -0.93 - 0.21(SO_2) \quad (5)$$

$r^2 = 0.84$, $p < 0.001$, 10 data points for beans only
($y = -0,93 - 0,60(SO_2)$, SO_2 in ppb)

The background mean SO_2 concentrations that provided the control levels in this study were low (7 - 9 µg m^{-3}, about 3 ppb). It is considered that the functions derived from this data, unlike those given by Baker *et al* (1986), may thus be applied directly without the need to consider how best to extrapolate back to 0 ppb SO_2. Therefore a negative effect of increased SO_2 on growth is anticipated by these functions at all concentrations above zero.

Functions 1 and 2 were used to quantify change in crop yield for wheat, barley, potato and sugar beet. Function 4 has been applied to barley only. Specific account was not taken of interactions with insect pests, climate etc. It is to be hoped that these elements are implicitly accounted for in the work by Baker *et al* because of the open air design of the experimental system.

It seems unlikely that plants with a high sulphur demand (e.g. rape, cabbage) would be adversely affected at current rural SO_2 levels as they should be able to metabolise and de-toxify any SO_2 absorbed.

2.2 O_3 Effects

Complete details of the assessment of ozone damages under the ExternE Project are given elsewhere (European Commission, 1998). In the same report alternative exposure-response functions are given in the chapter on ecological impact assessment, these being derived from European analysis (those given below are from work conducted in the USA). These are to be preferred for future analysis but were unavailable at the time that the ozone damage estimates were made for ExternE National Implementation.

A large number of laboratory experiments have clearly established that ozone, at concentrations commonly found in urban environments, has harmful effects on many plants. Exposure-response functions have been derived for several plants of economic importance. Nonetheless the quantification of crop damages is problematic. Laboratory experiments are typically carried out under very limited conditions (single species, single pollutant, particular exposure scenarios, controlled climate, etc.), and one wonders to what extent they are representative of real growing conditions in a variety of countries and climates. As an example of possible complexities see Nussbaum *et al* (1995) who subjected a mixture of perennial rye grass and white clover to several different ozone exposure patterns in the typical open-top chamber arrangement. This combination of plants was chosen because of their importance for managed pastures in Europe. The authors found that the ozone damage depended not only on the total exposure but also on the exposure pattern. Furthermore they found two thresholds: species composition is fairly well correlated with AOT40 (accumulated concentration of O_3 above 40 ppb in ppb.hours) but total forage yield with AOT110 (accumulated concentration of O_3 above 110 ppb).

Experiments in the USA derived a number of functions for different crops based on the Weibull function:

$$y_r = a \cdot e^{-(x/s)^c} \qquad (6)$$

where
y_r = crop yield,
a = hypothetical yield at 0 ppm ozone, usually normalised to 1,
x = a measure of ozone concentration,
s = ozone concentration when yield = 0.37,
c = dimensionless exponential loss function to reflect sensitivity.

The values derived experimentally for these parameters for different crops are shown in Table 1.

Table 1. Weibull function parameters for different crop species based on studies carried out under the NCLAN programme. s = ozone concentration when yield = 0.37, c = dimensionless exponential loss function to reflect sensitivity. The relevant ozone exposure metric is in ppb expressed as the seasonal 7 or 12 hour/day mean. All functions shown were derived using US data. Figures in parentheses denote approximate standard errors.

Crop	O_3 metric	s	c	Source
Alfalfa	12 hr/day	178 (2.8)	2.07 (0.55)	Somerville *et al*, 1989
Barley	no response			Somerville *et al*, 1989
Corn (*Zea mays*)	12 hr/day	124 (0.2)	2.83 (0.23)	Somerville *et al*, 1989
Cotton	12 hr/day	111 (0.5)	2.06 (0.33)	Somerville *et al*, 1989
Forage grass	12 hr/day	139 (1.5)	1.95 (0.56)	Somerville *et al*, 1989
Kidney bean	7 hr/day	279 (7.9)	1.35 (0.70)	Somerville *et al*, 1989
Soybean	12 hr/day	107 (0.3)	1.58 (0.16)	Somerville *et al*, 1989
Wheat	7 hr/day	136 (0.6)	2.56 (0.41)	Somerville *et al*, 1989
Sugar beet, turnip*	7 hr/day	94	2.905	Fuhrer *et al*, 1989
Spinach*	7 hr/day	135	2.08	Fuhrer *et al*, 1989
Lettuce*	7 hr/day	122	8.837	Fuhrer *et al*, 1989
Tomato*	7 hr/day	142	2.369	Fuhrer *et al*, 1989

Here we are concerned with marginal changes around current concentration values. Thus we consider the reduction in crop yield

$$\text{reduction in yield per ppb} = \frac{1}{y}\frac{dy}{d\text{Conc}} \tag{7}$$

relative to current agricultural production.

Exposure-response functions describing the action of ozone on crops have recently been developed using European data (Skärby *et al*, 1994). However, only 3 crops were covered, spring wheat, oats and barley, though the last 2 of these were found to be insensitive to O_3. An expert panel on crop damage convened under the ExternE Project concluded that rye was also unlikely to be sensitive to O_3. The following function was derived for sensitive crops;

$$Y_{rel} = 1 + 0.0008 \cdot x_8 - 0.000075 \cdot x_8^{\,2} \tag{8}$$

Where Y_{rel} = relative yield
x_8 = average daily peak 8 hour concentration.

The functions shown here refer to peak concentrations during 7, 8 or 12 hr periods. Ozone related crop damages were assessed against 6 hour peak values reported by Simpson (1992; 1993), generated from the EMEP model (Eliasson and Saltbones, 1983; Simpson, 1992). This model extends to the whole of Europe with a resolution of 150 km by 150 km. In addition the Harwell Global Ozone model (Hough, 1989; 1991) was also used, extending the zone of analysis to the whole of the Northern Hemisphere, though with greater uncertainty compared to the European analysis.

The various functions shown in this Appendix were used to generate an average function for crop loss (Table 2).

Table 2. Average and standard deviation of yield reduction for species in Fig.1 at Conc = 56 ppb. The first line shows a derivative of the Weibull function according to Equation 6, whilst the second line is the slope of the straight line from the origin to the value of the exposure-response function at 56 ppb.

	Average	**Standard Deviation**
Error! Error! from d-r function	-0.0058	0.0033
(y - 1)/Conc straight line	-0.0025	0.0014

For agriculture there is relatively little uncertainty and controversy at the valuation stage since we are dealing with market goods. Our analysis applied exposure-response functions directly to the total market volume of each crop. The omission of data on forage grass production for raising cattle is thought to be of little importance as the dominant species of forage grass, rye grass, is quite insensitive to ozone.

2.3 Acidification of Agricultural Soils

Soil acidification is seen as one of the major current threats to soils in northern Europe. It is a process which occurs naturally at rates which depend on the type of vegetation, soil parent material, and climate. Human activities can accelerate the rate of soil acidification, by a variety of means, such as the planting of certain tree species, the use of fertilisers, and by the draining of soils. However, the major concern in Europe is the acceleration of soil acidification caused by inputs of oxides of sulphur and nitrogen produced by the burning of fossil fuels.

UK TERG (1988) concluded that the threat of acid deposition to soils of managed agricultural systems should be minimal, since management practices (liming) counteract acidification and often override many functions normally performed by soil organisms. They suggested that the only agricultural systems in the UK that are currently under threat from soil acidification are semi-natural grasslands used for grazing, especially in upland areas. Particular concern has been expressed since the 1970’s when traditional liming practices were cut back or ceased altogether, even in some sensitive areas, following the withdrawal of government subsidies. Concern has also been expressed in other countries. Agricultural liming applications decreased by about 40% in Sweden between 1982 and 1988 (Swedish EPA, 1990). Although liming may eliminate the possibility of soil degradation by acidic deposition in well-managed land, the efficacy of applied lime may be reduced, and application rates may need to be increased.

The basis of the method is to calculate:

- The total amount of acidifying pollutant deposited to the land surface in a given area;
- The amount which falls on soils which require lime (excluding, for example, urban areas, water and soils on calcareous drifts);
- The cost of neutralising this amount of acidic deposition with lime;

- The increased acidic deposition in this area resulting from the operation of a fuel cycle;
- The additional cost of neutralising the increase in inputs to soils which require lime.

Deposition values for acidity are typically expressed in terms of kilo-equivalents (keq) or mega-equivalents (Meq). One equivalent is the weight of a substance which combines with, or releases, one gram (one equivalent) of hydrogen. When sulphuric acid is neutralised by lime (calcium carbonate);

$$H_2SO_4 + CaCO_3 \rightarrow CaSO_4 + H_2O + CO_2$$

The following conversion factors were used in our analysis;

1 keq/ha = 10^4 keq/10 km square, or 10 Meq/10 km square.

Also, 1 Meq/100 km square = 1 eq/ha.

Each grid square within the study area is given a weighting (between 0 and 1) for the proportion of the square that would need to be limed (excluding urban areas, water and soils on calcareous drifts). So far, analysis has only been conducted for the UK, and the source of data at this point was the Soil Survey of England and Wales 1:1,000,000 Soil Map of England and Wales. The total acidifying pollution input multiplied by the soil weighting factor gave the total acidifying pollution input per unit area on soils which require lime.

100 kg $CaCO_3$ is sufficient to neutralise 2 kg H^+. Accordingly the total acidifying pollution input on soils which require lime was multiplied by 50 to give the amount of lime which required to neutralise it. As the total acidifying pollution input was in eq, the amount of lime is in Mg (tonnes). One tonne per 100 km sq is equivalent to 1g/ha on average. Hence, the amount of lime in t/100 km sq is divided by 1000 to give the mean kg/ha. The price of lime is ECU 16.8/tonne.

2.4 Fertilisational Effects of 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 a farmer is not excessive). The analysis is conducted in the same way as assessment of effects of acidic deposition. The benefit is calculated directly from the cost of nitrate fertiliser, ECU 430/tonne of nitrogen (note: not per tonne of nitrate) (Nix, 1990). Given that additional inputs will still be needed under current conditions to meet crop N requirements there is a negligible saving in the time required for fertiliser application (if any).

3. Modelling Air Pollution Damage to Forests

3.1 Introduction

Forest growth models are made particularly complex by the fact that trees are long lived and need to be managed sustainably. To ensure an adequate supply of timber in future years it is thus important that harvests are properly planned. Even under ideal conditions harvesting levels cannot be suddenly increased beyond a point at which the amount of standing timber starts to fall, without either reducing the amount of timber cut in future years or requiring rapid expansion of the growing stock. If acidic deposition has serious effects on tree growth (which seems likely) it is probable that impacts associated with soil acidification will persist for many years after soils have recovered, whilst the quantity of standing timber recovers to a long term sustainable level.

The following modelling exercises were reviewed in an earlier phase of the study (European Commission, 1995a, Chapter 9):

- NAPAP (the US National Acid Precipitation Assessment Program) review (Kiester, 1991);
- The IIASA Forest Study Model (Nilsson *et al*, 1991; 1992);
- The forest module of the RAINS model (Makela and Schopp, 1990).

In the NAPAP review Kiester (1991) concluded that;

'*None of the models can now be used to produce precise quantitative projections because of uncertainties in our understanding of key growth processes and lack of adequate data sets.'*

Although the work of Nilsson *et al* and Makela and Schopp provided useful insights into forest damage issues, neither study was regarded as being widely applicable. In addition, serious questions were raised regarding the form of the model derived by Nilsson.

In the absence of directly applicable models for assessment of the effects of fuel cycle emissions on forests, further work, some of it conducted as part of the ExternE Project (European Commission, 1995a, Chapter 9), has sought to develop novel approaches to the assessment of forest damage in the last few years. As yet, none of these approaches have been properly validated, and results vary widely.

The first stage in most approaches concerns the identification of sensitive areas through analysis of critical loads and levels, taking account of variation in soil conditions and the type of ecosystem present. These maps are then be integrated with deposition maps (accounting for acidifying, neutralising and eutrophying inputs) to identify the areas where critical loads and levels are exceeded, and hence where the underlying ecosystems are at risk.

3.2 Sverdrup and Warvinge (1993)

Sverdrup and Warfvinge (1993) developed an approach based on Ulrich's hypothesis (1985, 1990). This hypothesis rests on the concept that, as soils acidify, aluminium will be released into the soil solution, and that mobile soil aluminium levels become high enough to case root damage. Well buffered ecosystems are unlikely to be affected by commonly occurring rates of acidic deposition in Europe. Sites that are already acidic will respond to acidic deposition by increases in aluminium concentrations and will therefore be more sensitive.

Sverdrup and Warfvinge's approach relates the ratio of base cations (Ca+Mg+K, which moderate the effects of aluminium) to Al in the soil solution, to tree growth. With regards to the mechanism the paper concentrates almost exclusively on concepts relating to ion uptake. It has been criticised (Kuylenstierna and Chadwick, 1994) for not considering other important conceptual issues, and for ignoring much of the considerable body of early literature which relates species distribution to soil acidity. In particular, concern was noted about the lack of agreement between the assessment of relative tolerance to acidity by Sverdrup and Warfvinge and ecological work on species distribution. Kuylenstierna and Chadwick stated that there is a need for further work to integrate ecological observation into the assessment of relative tolerance.

An economic assessment of damage using this type of approach was made by Sverdrup *et al* (1993). Estimates of damage were substantial, though restricted to effects in Norway and Sweden. This approach, as it currently stands, therefore suggests that the externalities of acidifying emissions on forest productivity are likely to be significant.

3.3 Kuylenstierna and Chadwick (1994)

The analysis presented by Kuylenstierna and Chadwick is, like Sverdrup and Warfvinge's work, based on Ulrich's hypothesis. The relationship between critical loads exceedance and acidification of the soil provides a potential cause-effect relationship with forest damage.

The approach goes through the following stages:
1. Quantification of an area weighted average critical loads exceedance for each country;
2. Assessment of the effects of critical loads exceedance on forest defoliation, using data on the health of forests collated by the UN ECE (GEMS, 1988-1991);
3. Assessment of the effects of defoliation induced by critical loads exceedance on timber production (based on results from Söderberg, 1992).

This approach was only intended as a first attempt to investigate whether data on forest condition collected throughout Europe can be integrated into a simple model of forest response to acidic deposition. An improved analysis would assess relationships between critical loads exceedance and crown condition at a finer level of detail than national data. Another obvious improvement would be the inclusion of additional factors in the relationships used to reduce the high degree of scatter that is currently present - this would become possible once the analysis was brought down to a finer resolution. Use of a finer resolution would also permit checking of results against data collected in the field.

It is necessary to ask what, exactly, the results of this approach tell us. A particular criticism is the fact that explicit account is not taken of the time frame over which a given amount of deposition will affect an ecosystem. Essentially the approach assumes that crown condition in any year is a consequence of critical loads exceedance in that year - a very rough approximation. Overall, because of the failure to account for other sources of variability in the national data on exceedance and defoliation, the approach is likely to underestimate impacts.

In spite of these reservations, this approach has been applied in some ExternE implementations. It was preferred to other methods because it uses data from field observations collected on a consistent basis throughout Europe, the simplicity of the approach, and the potential for further development work. Further to this, like the work of Sverdrup and Warfvinge, the approach highlights the fact that there is benefit to be gained from examination of the wealth of data that is available for assessment of forest damage.

Different approaches have also been used in the ExternE National Implementation Programme (see next section). This emphasises the fact that, at the present time, there is no single approach for dealing with forest damage that is obviously better than any other.

3.4 Use of the critical loads and levels concept with exposure-response functions specific to German forests

This analysis also starts from identification of sensitive areas through the application of the concept of critical loads and levels. In a German case study it proceeded as follows;

- The correlation between forest damage and pollutant burdens was tested (FBWL, 1989, based on the percentage of Norway spruce in damage classes 2-4 in different forest areas in 1986 was used. This was correlated with:
    - Total deposition of sulphate and nitrate (throughfall) from 1983 to 1987; and

* Mean 7 h/day ozone concentration during summer (15.4. - 16.10), averaged from 1984 to 1988 (only rural measurement stations with at least three years of measurement have been used).

The participants of an international forest damage workshop convened under this study in May 1992 agreed that this correlation could be used to estimate forest damage for German Norway spruce forests, though they felt that areas at higher altitudes should be excluded.

Another correlation between forest damage inventory and ozone concentration is provided by Kley *et al* (1990). Damage classes 2-4 of the Forest Damage Inventory 1986 were again used. These were correlated with the 7 hour per day mean ozone concentration over the period 1984 - 1988 during summer in two steps. First, different forest regions were grouped according to area damaged and the ozone concentration for the forest regions in each group were averaged. Secondly, the correlation between the percentage of damaged forest area and the averaged ozone concentration was calculated. A value for r of 0.81 ($p < 0.025$) was calculated from the six data points available. Kley *et al* emphasised that the standard deviation of the mean values is high.

Of course, a statistical relation is not synonymous with proof of causality but an extension of these correlation examples could be a way to develop a rough estimation model for forest damage at least for Germany. In the meantime the data base has improved as two forest damage inventories (1986 and 1991) with a spatial resolution as small as forest region have become available.

Like other approaches identified above, a number of limitations and uncertainties apply to the methodology defined in this section:

- It is restricted to West German Norway spruce forest below 800 m (about 37 % of the West German forests);
- There is a high uncertainty associated with the factors for the filtering capacity of the canopy;
- Characterisation of the response (damage classes of the Forest Damage Inventory) is unspecific for the different decline types;
- There are no relationships available for other influencing factors such as nitrogen enrichment, climate, forestry measures, etc;
- Forms of damage other than defoliation are not considered.

3.5 Cost analysis based on the application of mitigating measures

Kroth *et al* (1989) assessed the silvicultural measures which forest managers apply to counteract forest damages, and associated costs for Germany. Table 3 shows the total forest areas to which these measures have been applied in West Germany in the period 1988 to 1992. Using the specific costs Kroth *et al* calculated totals for the whole of West Germany. In the table only those measures which have been approved by experts to have mitigating potential and which are separable from normal operation have been listed. Thus, the total costs for West Germany are in the range of 41.2 to 112.9 MECU/year for the five year period.

Table 3. Costs of mitigating measures and forest areas totally affected in West Germany between 1988 and 1992, from Kroth *et al*.

Mitigating measure	Forest area used - 1988 to 1992 (1000 ha)	Costs (ECU/ha)	Total costs for 1988 to 1992 (MECU/year)
Liming and supplementary fertilisation	28 - 230	150 - 500	6.85 - 51.3
Site mapping	344	33.5 - 35	2.4
Reforestation of damaged stands	0.3	0 - 5,000	4.95
Remodelling	1.7 - 4.0	2,500 - 10,000	14.6 - 32.0
Cultivation of underwood	2.4 - 6.0	600 - 5,100	5.3 - 12.9
Renovation of protection forests	0.16	mean 30,000	4.8

Biological protection of forests	not quantified	/	1.75
Reduction of deer	not quantified	/	0.5 - 2.75 (only state forests)
Total			41.2 - 112.9

The total figure can be divided by the total area of spruce forest in West Germany subject to critical loads or levels exceedance to provide an estimate of cost per hectare over a five year period. Multiplying this by the incremental increase in area under critical loads and levels exceedance due to operation of the fuel cycle provides a lower estimate of damages, assuming that such measures would be applied. The assessment provides a lower boundary because the analysis is, at the present time, incomplete.

3.6 Summary of alternative approaches identified in this study

All of the approaches are linked to assessment of critical loads exceedance. The correlation between critical load exceedance by acidic deposition associated with sulphur and soil pH in Europe shows that this initial condition in the cause-effect relationship holds.

There are indications that defoliation, particularly amongst conifers is related to the rate of acidic deposition and critical load exceedance. Defoliation is of limited interest as a response variable and therefore the relationships between defoliation and increment suggested by the work of Söderberg (1992) are of much significance.

The methodologies presented here are acknowledged to be preliminary. They demonstrate that a link between probable cause (critical loads exceedance for acidity) and effect (reduced timber growth) can be traced using simple data, rather than complex models. It is hoped that this analysis will stir the debate in this area, and identify refinements to the approach (or alternatives) that will allow analysis to be performed with greater confidence.

4. Assessment of Nitrogen Deposition and Critical Loads Exceedence

?Petra - do you have any suitable text for this?

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APPENDIX V: ASSESSMENT OF GLOBAL WARMING DAMAGES

1. Introduction

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

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

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

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

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

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

- discount rate
- the treatment of equity,
- the value of statistical life, and
- the magnitude of higher order effects.

These issues have now been explored in more depth using two models - FUND, developed by Richard Tol of the Institute for Environmental Studies at the Vrije Universiteit in Amsterdam, and the Open Framework, developed by Tom Downing and colleagues at the Environmental Change Unit at the University of Oxford. So far as is reasonable, the assumptions within the FUND and Open Framework models are both explicit and consistent. However, the models are very different in structure and purpose, so that convergence is neither possible nor desirable. Another major advantage over previous work is that the models both enable specific account to be taken of the scenarios developed by IPCC. Further details are provided by the ExternE Project report on climate change damage assessment (European Commission, 1998).

Numerous impacts are included in the two models, ranging from effects on agricultural production to effects on energy demand. Details of precisely what is included and excluded by the two models is provided by European Commission (1998).

2. Interpretation of Results

Section 4 of this appendix contains selected results for the base case and some sensitivity analyses. The results given have been selected to provide illustration of the issues that affect the analysis - they are not a complete report of the output of the ExternE global warming task team.

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

In view of these problems, and in the interests of providing policy makers with good guidance, the task team has sought (though inevitably within limits) to avoid introducing personal bias on issues like discount rate, which could force policy in a particular direction. There is a need for other users of the results, such as energy systems modellers or policy makers, to both understand and pass on information regarding uncertainty, and not to ignore it because of the problems that inevitably arise. The Project team feel so strongly about this that reference should not be made to the ExternE Project results unless reference is also made to the uncertainties inherent in any analysis and our attempts to address them. Reliance on any single number in a policy-related context will provide answers that are considerably less robust than results based on the range, although this, in itself, is uncertain.

3. Discounting Damages Over Protracted Timescales

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

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

It is necessary to look in more detail at the consequences of using different discount rates for analysis of damages that occur in the long term future (Figure 1). A rate of 10% (typical of that used in commercial decision making) leads after only 25 years to damages falling to a negligible level (taken here for illustration as being less than 10% of the original damages). For a 3% discount rate this point is reached after 77 years. For 1% it is reached after 230 years. The use of a rate of 10% clearly looks inappropriate from the perspective of soft-sustainability to which the European Union is committed, given long term growth rates. However, the choice between 3% and 1% on grounds of soft-sustainability is not so clear.

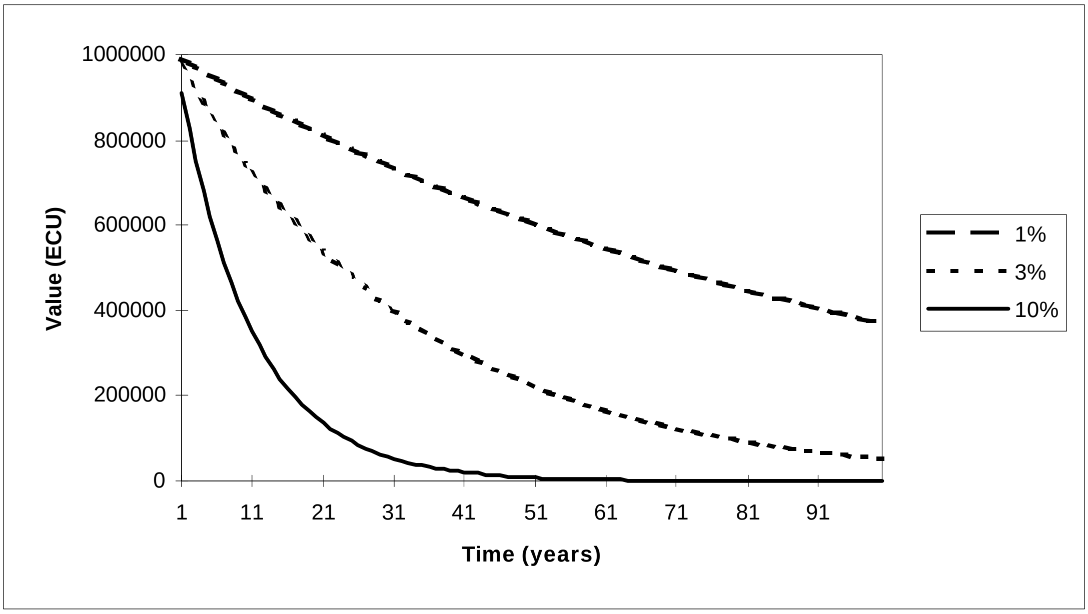

Figure 1 Effect of discount rate on present value of damages worth 1 million ECU at the time (from 1 to 100 years in the future) when damage is incurred.

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

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

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

4. Results

Damages have been calculated for a range of different assumptions using both models. For the base case results shown in Table 1 the overall marginal damages calculated by the two models are in good agreement. However, this does not reflect variation in damage estimates disaggregated to individual impact categories, such as agriculture and energy demand. As differences do exist in the disaggregated figures, the close agreement between the overall estimates could be regarded as largely fortuitous.

Table 1 Marginal damages (1990 $) of greenhouse gas emissions. A discount rate of 1% is used for the purposes of illustration only.

Greenhouse Gas	Damage Unit	Marginal Damage from Model	
		FUND	Open Framework
Carbon Dioxide, CO_2	$/tC	170	160
Methane, CH_4	$/t$CH_4$	520	400
Nitrous Oxide, N_2O	$/t$N_2O$	17,000	26,000

Source: FUND and Open Framework
Basis: 1% discount rate
IPCC IS92a scenario
equity weighted
no higher order effects
emissions in 1995-2005
time horizon of damages 2100

Data in Table 1 are quoted in 1990 US dollars, which is the norm for climate change damage work. For the purposes of ExternE, 1995 ECU is the standard currency and 1995 is the date at which the net present value of future damages are measured. The following conversion factors therefore need to be applied:

- 1990 ECU:1990 US$ currency conversion - a factor of 0.8,

- 1995 ECU: 1990 ECU consumer price index inflation - a factor of 1.2, and
- revaluation for a 1995 start year - a factor of 1.05 at a 1% discount rate, 1.15 at 3%.

The combined numerical effect of all these changes is a factor almost exactly equal to unity for a 1% discount rate, 1.1 for a 3% discount rate, and 1.2 for a 5% discount rate. The converted base case results at the 1% discount rate are presented in Table 2.

Table 2 Marginal damages (1995 ECU) of greenhouse gas emissions. A discount rate of 1% is again used for the purposes of illustration only.

Greenhouse Gas	Damage Unit	Marginal Damage from Model	
		FUND	Open Framework
Carbon Dioxide, CO_2	ECU/tC	170	160
Methane, CH_4	ECU/tCH_4	520	400
Nitrous Oxide, N_2O	ECU/tN_2O	17,000	26,000

Source: FUND and Open Framework
Basis: 1% discount rate
IPCC IS92a scenario
equity weighted
no higher order effects
emissions in 1995-2005
time horizon of damages 2100

This assessment has sought to make clear the effects of different assumptions on the marginal damages of climate change. The base case values for carbon dioxide damages calculated from the two models should not therefore be quoted out of context or taken to be a ‘correct’ value. Uncertainty analysis in FUND indicates a geometric standard deviation of approximately 1.8, for uncertainties in climate and impacts which can be parameterised. But many important issues cannot and create additional uncertainty. The treatment of equity, discount rate and possible higher order impacts in particular can have a large effect on damages. The effects of some of these sensitivities on the marginal damages of carbon dioxide (calculated in FUND only) are shown in Table 3. Assumptions about higher order effects could affect the results even more.

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

Table 3 FUND sensitivity analysis of marginal damages for CO_2 emissions.

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

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

5. Conclusions

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

The following ranges of estimates are recommended for use within the ExternE National Implementation Study (Table 4). It is stressed that the outer range derived is indicative rather than statistical, and is likely to underestimate the true uncertainty. The inner range is composed of the base-case estimates for the 1 and 3% discount rates, and is referred to here as the 'illustrative restricted range'. There was some debate as to whether the lower bound of this range should be reduced to take account of the 5% discount rate (which would have given a figure of [1995]ECU 8.8/tCO_2) but there was very limited support from the task team for use of the 5% rate. However, the 5% rate was used in derivation of the outer range.

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

Table 4 Recommended global warming damage estimates for use in the ExternE National Implementation Study. The ranges given do not fully account for uncertainty. The derivation of each of the figures identified is described in the text.

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

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APPENDIX VI: FURTHER ANALYTICAL DETAILS

Table VI.1 Human health exposure-response functions - Base analysis

Receptor	Impact	Reference	Pollutants
ASTHMATICS			
adults	Bronchodilator usage	Dusseldorp *et al.*, 1995	PM_{10}, Nit, Sul
	Cough	Dusseldorp *et al.*, 1995	PM_{10}, Nit, Sul
	Lower respiratory symptoms (wheeze)	Dusseldorp *et al.*, 1995	PM_{10}, Nit, Sul
children	Bronchodilator usage	Roemer *et al.*, 1993	PM_{10}, Nit, Sul
	Cough	Pope and Dockery, 1992	PM_{10}, Nit, Sul
	Lower respiratory symptoms (wheeze)	Roemer *et al.*, 1993	PM_{10}, Nit, Sul
ELDERLY (above 65 years old)			
	Congestive heart failure	Schwartz and Morris, 1995	PM_{10}, Nit, Sul
CHILDREN			
	Chronic bronchitis	Dockery *et al.*, 1989	PM_{10}, Nit, Sul
	Chronic cough	Dockery *et al.*, 1989	PM_{10}, Nit, Sul
ADULTS			
	Restricted activity days	Ostro, 1987	PM_{10}, Nit, Sul
	Chronic bronchitis	Abbey *et al.*, 1995	PM_{10}, Nit, Sul
	'Chronic' YOLL	Pope *et al.*, 1995	PM_{10}, Nit, Sul
ENTIRE POPULATION			
	Respiratory hospital admissions	Dab *et al.*, 1996	PM_{10}, Nit, Sul
	Respiratory hospital admissions	Ponce de Leon, 1996	SO_2
	Cerebrovascular hospital admissions	Wordley *et al.*, 1997	PM_{10}, Nit, Sul
	'Acute' YOLL	Spix *et al.*, 1996, Verhoeff *et al.*, 1996	PM_{10}, Nit, Sul
	'Acute' YOLL	Anderson *et al.*, 1996, Touloumi *et al.*, 1996	SO_2

Table VI.2 Human health - Sensitivity analysis 1: value of statystical life for acute mortality is used rather than years of life lost approach

Receptor	Impact	Reference	Pollutants
ENTIRE POPULATION			
	Acute mortality	Spix *et al.*, 1996, Verhoeff *et al.*, 1996	PM_{10}, Nit, Sul
	Acute mortality	Anderson *et al.*, 1996, Touloumi *et al.*, 1996	SO_2

Table VI.3 Human health - Sensitivity analysis 2: value of statystical life for chronic mortality is used rather than years of life lost approach

Receptor	Impact	Reference	Pollutants
ADULTS			
	Chronic mortality	Pope *et al.*, 1995	PM_{10}, Nit, Sul

Table VI.4 Human health - Sensitivity analysis 3: acute mortality from SO_2 is neglected

Receptor	Impact	Reference	Pollutants
ENTIRE POPULATION			
	Acute mortality	Anderson *et al.*, 1996, Touloumi *et al.*, 1996	SO_2

Table VI.5 Human health - Sensitivity analysis 4: acute mortality from NO_x is included

Receptor	Impact Category	Reference	Pollutant
ENTIRE POPULATION			
	'Acute' YOLL	Sunyer *et al.*, 1996, Anderson *et al.*, 1996	NO_x

Table VI.6 Human health - Sensitivity analysis 5: particulates are treated as $PM_{2.5}$ rather than PM_{10}

Receptor	Impact	Reference	Pollutants
ASTHMATICS			
adults	Bronchodilator usage	Dusseldorp *et al.*, 1995	$PM_{2.5}$, Nit, Sul
	Cough	Dusseldorp *et al.*, 1995	$PM_{2.5}$, Nit, Sul
	Lower respiratory symptoms (wheeze)	Dusseldorp *et al.*, 1995	$PM_{2.5}$, Nit, Sul
children	Bronchodilator usage	Roemer *et al.*, 1993	$PM_{2.5}$, Nit, Sul
	Cough	Pope and Dockery, 1992	$PM_{2.5}$, Nit, Sul
	Lower respiratory symptoms (wheeze)	Roemer *et al*., 1993	$PM_{2.5}$, Nit, Sul
ELDERLY (above 65 years old)			
	Congestive heart failure	Schwartz and Morris, 1995	$PM_{2.5}$, Nit, Sul
CHILDREN			
	Chronic bronchitis	Dockery *et al.*, 1989	$PM_{2.5}$, Nit, Sul
	Chronic cough	Dockery *et al.*, 1989	$PM_{2.5}$, Nit, Sul
ADULTS			
	Restricted activity days	Ostro, 1987	$PM_{2.5}$, Nit, Sul
	Chronic bronchitis	Abbey *et al.*, 1995	$PM_{2.5}$, Nit, Sul
	'Chronic' YOLL	Pope *et al.*, 1995	$PM_{2.5}$, Nit, Sul
ENTIRE POPULATION			
	Respiratory hospital admissions	Dab *et al.*, 1996	$PM_{2.5}$, Nit, Sul
	Respiratory hospital admissions	Ponce de Leon, 1996	SO_2
	Cerebrovascular hospital admissions	Wordley *et al.*, 1997	$PM_{2.5}$, Nit, Sul
	'Acute' YOLL	Spix *et al.*, 1996, Verhoeff *et al.*, 1996	$PM_{2.5}$, Nit, Sul
	'Acute' YOLL	Anderson *et al.*, 1996, Touloumi *et al.*, 1996	SO_2

Table VI.7 Human health - Sensitivity analysis 6: some more exposure response functions are included

Receptor	Impact Category	Reference	Pollutant
ELDERLY (above 65 years old)			
	Ischaemic heart disease	Schwartz and Morris, 1995	PM_{10}, Nit, Sul
ENTIRE POPULATION			
	Respiratory hospital admissions	Ponce de Leon, 1996	NO_x
	ERV for COPD	Sunyer *et al.*, 1993	PM_{10}, Nit, Sul
	ERV for asthma	Schwartz, 1993 and Bates, 1990	PM_{10}, Nit, Sul
	Hosp. visits child. croup	Schwartz *et al.*, 1991	PM_{10}, Nit, Sul
ADULTS			
	Asthma	Abbey	Sulfates
FEMALES			
	Malignant Neoplasms	Abbey	Sulfates

Table VI.8 Human health - Sensitivity analysis 7: some exposure response functions are neglected

Receptor	Impact	Reference	Pollutants
ADULTS			
	Restricted activity days	Ostro, 1987	PM_{10}, Nit, Sul
ENTIRE POPULATION			
	Respiratory hospital admissions	Ponce de Leon, 1996	SO_2

APPENDIX VII: VALUATION ISSUES

1. Introduction

The purpose of this Appendix is to provide additional background material relevant to the valuation of the impacts that have been quantified using the techniques described above. Little detail is provided here - this Appendix is not intended to provide any more than a brief introduction to the general methods employed in environmental economics. Some issues are dealt with in more depth in other Appendices, such as Appendix II which dealt with analysis of health damages. More complete details are provided in the ExternE Methodology Reports (European Commission, 1995; 1998).

The following issues are covered;

- Techniques for eliciting the value of goods and services
- Categories of value
- Transferability of valuation data
- Estimation of uncertain and risky phenomena
- Discounting

2. Techniques

Valuation data for energy externalities studies need to be derived from a number of sources. Over the last 25 years or so, a number of techniques have been developed for estimating external environmental effects. A survey of these may be found in Pearce *et al* (1989).

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

A good example to start with concerns changes in crop yield. In this case market prices are a reasonable metric for damage assessment, although even in this simple case there are problems and issues that arise (see European Commission, 1995, pp 455-459). For a wide range of impacts, however, such as increased risk of death or loss of recreational values, there are no direct market prices that can be used. Three techniques are widely used in this context. One is elicitation of the WTP or WTA by direct questionnaire. This is termed the *contingent valuation method* and is widely applicable. Another is to consider how the WTP is expressed in related markets. An increase in noise or a reduction in visibility (all other things being equal) tends to lead to a reduction in the value of affected properties. This approach is called the *hedonic price method* and is widely used for noise and aesthetic effects.

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

3. Categories of Value

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

There are many different categories of use value. Direct use values arise when an individual makes use of the environment (e.g. from breathing the air) and derives a loss of welfare if that environment is polluted. Indirect use values arise when an individual's welfare changes in response to effects on other individuals, for example, in response to the death or illness of a friend or relation. This can and has been measured in limited cases and is referred to as an altruistic value.

Another category of use value that is potentially important is that of option value. This arises when an action taken now can result in a change in the supply or availability of some environmental good in the future. For example, as a consequence of flooding a region to impound water for a hydro project. People might have a WTP for the option to use the area for hiking or some other activity, even if they were not sure that it would ever be used. This WTP is the sum of the expected gain in welfare from the use of the area, plus a certain gain in welfare from the knowledge that it could be used, even if it is not already. The latter is referred to as the option value. The literature on environmental valuation shows that, in certain cases the option value will be positive but in general it is not an important category of value, and hence has been excluded from the ExternE study.

The last category of value is non-use value. This is a controversial area, although values deriving from the existence of a pristine environment are real enough, even for those who will never make any use of it. In some respects what constitutes 'use' and what constitutes 'non-use' is not clear. Pure non-use value must not involve any welfare from any sensory experience related to the item being valued. In fact some environmentalists argue that such non-use or existence values are unrelated to human appreciation or otherwise of the environment, but are embedded in, or intrinsic to, the things being valued. However, the basis of valuation in this study is an anthropocentric one which, however many economists argue, does not imply an anti-environment stance.

The difficulty in defining non-use values extends to measuring them. The only method available is contingent valuation (see above). This method has been tested and improved extensively in the past 20 years. The general consensus is that the technique works effectively where 'market conditions' of exchange can reasonably be simulated and where the respondent has considerable familiarity with the item being valued. For most categories of non-use value this is simply not the case. Hence, for the present, non-use values are extremely difficult to value with any accuracy and are not covered in this study.

4. Transferability of Valuation Data

4.1 Benefit Transfer

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

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

Benefit transfer application can be based on: (a) expert opinion, or (b) meta analysis, discussed below.

4.2 Expert Opinion

This is carried out by asking experts how reasonable it is to make a given transfer and then determining what modifications or proxies are needed to make the transfer more accurate. In many cases expert opinion has been resorted to in making the benefit transfer during the ExternE Project. More detailed comments on the issues involved in transferring the benefits were given in Section B of the original ExternE Valuation Report (European Commission, 1995, Part II). In general the more 'conditional' the original data estimates (e.g. damages per person, per unit of dispersed pollution, for a given age distribution) the better the benefit transfer will be. In one particular case (that of recreational benefits) an attempt was made to check on the accuracy of a benefit transfer by comparing the transferred damage estimate with that obtained by a direct study of the costs (see European Commission, 1995, Part II, Chapter 12). The finding there was not encouraging in that the two figures varied by a wide margin.

4.3 Meta Analysis

Meta analysis is performed by taking damages estimated from a range of studies and investigating how they vary systematically with the size of the affected population, building areas, crops, level of income of the population, etc. The analysis is carried out using econometric techniques, which yield estimates of the responsiveness of damages to the various factors that render them more transferable across situations.

4.4 Conclusions on benefit transfer

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

It is important to note that national boundaries themselves are not of any relevance in transferring estimates, except that there may be cultural differences that will influence factors such as frequency with which a person visits a doctor, or how he perceives a loss of visibility. In this sense there is no reason why a Project like ExternE should not draw on the non-European literature (particularly that from the USA)

5. Estimation of Uncertain and Risky Phenomena

A separate but equally important aspect of the uncertainty dimension in the valuation of environmental impacts arises from the fact that, for the health related damages, one is valuing changes in risk of damage. Thus the health impacts are usually in the from of an increased risk of premature death or of ill health at the individual level.

For health damages estimated in the form of increased likelihood of illness it is not sufficient to take the cost of an illness and multiply it by the probability of that illness occurring as a result of the emissions. The reasons are (a) that individuals place a considerable value on not experiencing pain and suffering (as do their friends and relations), and (b) individuals place a value on the *risk* itself.

Estimating the risk premium is very important, especially when it comes to environmental damages related to health. It can be assessed by using contingent valuation methods, or by looking at actual expenditures incurred to avert the impacts; it cannot be valued by looking at the cost of treatment alone. It is also important to note that the premium will depend not only on the shape of the utility function (which indicates attitudes to risk aversion), but also on the *perceived probabilities* of the damages. There is some evidence to indicate that, for events with small probabilities of occurrence, the subjective probabilities are often much higher than the objective ones.

Another aspect of the value of risk in the context of environmental problems is that individuals have very different WTA's for increased risk, depending on whether the risk is voluntarily incurred, or whether it is imposed from outside. Thus, the WTP to reduce the risk of health effects from air pollution will typically be much higher than the WTA payment to undertake a risky activity, such as working in an industry with a higher than average risk of occupational mortality and morbidity. The reasons for the higher values of involuntary risk are not altogether clear, but undoubtedly have something to do with perceived natural rights and freedom of choice. Since most of the estimated values of increased risk are taken from studies where the risk is voluntary, it is very likely to be an underestimate of the risk in an involuntary situation such as a nuclear accident.

6. Discounting

6.1 Introduction

Discounting is the practice of placing lower numerical values on future benefits and costs as compared to present benefits and costs. In the context of this study it is an important issue because many of the environmental damages of present actions will occur many years from now and the higher the discount rate, the lower the value that will be attached to these damages. This has already been illustrated in Appendix V, dealing with global warming damages and has major implications for policy.

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

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

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

Clearly the higher r and T are, the lower the value of the discounted damages. Typically discount rates in EC countries run at around 5 to 7 % in real terms. ['real terms' means that no allowance is made for general inflation in the computation of future values, and all damages are calculated in present prices.]

6.2 The Discounting Debate from an Environmental Perspective

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

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

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

6.2.1 Pure Individual Time Preference

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

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

6.2.2 Social Rate of Time Preference

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

$$i = ng + z$$

where z is the rate of pure time preference, 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’). A typical value for n would be one. With no growth in per capita consumption, the social rate of time preference would be equal to the private rate, z. If consumption is expected to grow the social rate rises above the private rate. The intuitive rationale here is that the more one expects to have in the future, the less one is willing to sacrifice today to obtain even more in the future. Moreover, this impact is greater the faster marginal utility falls with consumption.

Many commentators point to the *presumed* positive value of g in the social time preference rate formula. First, they argue that there are underlying ‘limits’ to the growth process. We cannot expect positive growth rates of, say, 2-3% for long periods into the future because of natural resource constraints or limits on the capacity of natural environments to act as ‘sinks’ for waste products. There are clearly some signs that the latter concern is one to be taken seriously, as with global warming from the emission of greenhouse gases and ozone layer depletion. But the practical relevance of the 'limits' arguments for economic planning is more controversial, although it may have more relevance for the *way* in which economies develop rather than for a reconsideration of the basic growth objective itself.

Assuming it is reasonable to use pure time preference rates at all, are such rates acceptable? In the context of developed countries there is little reason to question such rates as long as the underlying growth rates on which they are based are believed to be sustainable. If the present rate is not considered sustainable, a lower rate should be employed. Taking a low sustainable rate of around 1-2% in real per capita terms for the European Union and setting the pure time preference rate to zero on ethical grounds would give a social time preference discount rate of around 1-2% as well. This could rise by one or two percentage points if one allows for a pure time preference rate of that amount.

6.2.3 Opportunity Cost of Capital

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

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

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

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

6.2.4 Risk and Uncertainty

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

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

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

Second, uncertainty about preferences is relevant to certain goods and perhaps even certain aspects of environmental conservation. However, economists generally accept that the way to allow for uncertainty about preferences is to include *option value* in an estimate of the benefit or cost rather than to increase the discount rate.

The third kind of uncertainty is relevant, but the difficulty is in allowing for it by adjusting the discount rate. Such adjustments assume that the scale of risks is increasing exponentially over time. Since there is no reason to believe that the risk factor takes this particular form, it is inappropriate to correct for such risks by raising the discount rate. This argument is in fact accepted by economists, but the practice of using risk-adjusted discount rates is still quite common among policy makers.

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

6.2.5 The Interests of Future Generations

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

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

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

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

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

6.3 Discount Rates and Irreversible Damage

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

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

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

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

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

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

6.4 A Sustainability Approach

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

Arguments against the use of opportunity cost of capital discount rates were also, in general, not found to be persuasive. It was also observed that, to account for uncertainty in investment appraisal, it was better to adjust the cost and benefit streams for the uncertainty rather than to add a 'risk premium' onto the discount rate. Finally, under the general re-analysis of the rationale for discounting, the arguments for adjusting discount rates on various grounds of inter-generational justice were examined. Although many of these arguments have merit, it was concluded that adjusting the discount rate to allow for them was not, in general, a practicable or efficient procedure. However, the need to protect the interests of future generations remains paramount in the environmental critique of discounting. Some alternative policy is therefore required if the discount rate adjustment route is not to be followed. One approach is through a 'sustainability constraint'.

The sustainability concept implies that economic development requires a strong protective policy towards the natural resource base. In the developing world one justification for this would be the close dependence of major parts of the population on natural capital (soil, water and biomass). More generally, ecological science suggests that much natural capital cannot be substituted for by man-made capital (an example might be the ozone layer).

If conservation of natural environments is a condition of sustainability, and if sustainability meets many (perhaps all) of the valid criticisms of discounting, how might it be built into project appraisal? Requiring that no project should contribute to environmental deterioration would be absurd. But requiring that the overall *portfolio* of projects should not contribute to environmental deterioration is not absurd. One way to meet the sustainability condition is to require that any environmental damage be *compensated* by projects specifically designed to improve the environment. The sustainability approach has some interesting implications for project appraisal, one of these being that the problem of choice of discount rates largely disappears.

To some extent, a sustainability approach is already followed in some key cases where protection of key resources and environments is guaranteed, *irrespective* of whether it can be justified on cost-benefit grounds at conventional discount rates. Although there are merits in favour of such an argument, what is being called for here is more than that. What is needed is a *systematic procedure* by which a sustainability criterion can be invoked in support of certain actions. Such a procedure does not exist, but it would be desirable to develop one.

6.5 Conclusions

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

- the arguments against any discounting at all are not valid;
- a social time preference rate of around 2-4% would be justified on the grounds of incorporating a sustainable rate of per capita growth and an acceptable rate of time preference;
- rates of discount based on the opportunity cost of capital would lie at around 5-7% for EU countries. There are arguments to suggest that these may be too high on social grounds. It is important to note that these arguments are not specific to environmental problems;
- the treatment of uncertainty is better dealt with using other methods, than modifying the discount rate;

- where irreversible damages are incurred, it is better to allow for these by adjusting the values of future costs and benefits than by employing a lower discount rate specifically for that project or component;
- for projects where future damage is difficult to value, and where there could be a loss of natural resources with critical environmental functions, a 'sustainability' approach is recommended. This implies debiting the activity that is causing the damage with the full cost of repairing it, irrespective of whether the latter is justified.

For the ExternE study it was recommended that the lower time preference rate be employed for discounting future damages, and a figure of 3% was selected as an acceptable central rate. In addition, appropriate increases in future values of damages to allow for increased demands for environmental services in the face of a limited supply of such facilities, should be made. A range of rates from 0% to 10% was also recommended. The range obtained provides an indication of the sensitivity of damage estimation to discounting. It is acknowledged that a 10% rate is excessive, but has been applied simply to demonstrate the effect of discounting at commercial rates. In Appendix V the problems of discounting even at a rate of 3% were identified, primarily for global warming assessment, but also (and more clearly) in the case of assessment of damages linked to disposal of high level radioactive waste. In these cases a rate lower than 3% may be acceptable.

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APPENDIX VIII: UNCERTAINTY AND SENSITIVITY ANALYSIS

1. Introduction

In numerous places in this report it has been made clear that uncertainties in external costs analysis are significant. The best estimate of any damages value is therefore, on its own, inadequate for most policy making purposes. Some indication of the credibility of that estimate, the likely margin of error, and the assumptions which might lead to significantly different answers, is also required.

It is appropriate to group the main contributions to the uncertainty into qualitatively different categories:

- statistical uncertainty - deriving from technical and scientific studies, e.g. dose-response functions and results of valuation studies,
- model uncertainty - deriving from judgements about which models are the best to use, processes and areas excluded from them, extension of them to issues for which they are not calibrated or designed. Obvious examples are the use of models with and without thresholds, use of rural models for urban areas, neglecting areas outside dispersion models and transfer of dose-response and valuation results to other countries,
- uncertainty due to policy and ethical choices - deriving from essentially arbitrary decisions about contentious social, economic and political questions, for example decisions on discount rate and how to aggregate damages to population groups with different incomes and preferences,
- uncertainty about the future - deriving from assumptions which have to be made about future underlying trends in health, environmental protection, economic and social development, which affect damage calculations, e.g. the potential for reducing crop losses by the development of more resistant species, and
- human error.

For human error, little can be done other than by attempting to minimise it. The ExternE Project uses well reviewed results and models wherever available and calculations are checked. The use of standardised software (EcoSense) has greatly assisted this.

Uncertainties of the first type (statistical) are amenable to analysis treatment by statistical methods, allowing the calculation of formal confidence intervals around a best estimate. Uncertainties in the other categories are not amenable to this approach, because there is no sensible way of attaching probabilities to judgements, scenarios of the future, the ‘correctness’ of ethical choices or the chances of error. There is no reason to expect that a statistical distribution has any meaning when attempting to take into account the possible variability in these parameters. In addition, our best estimate in these cases may not be a median value, thus the uncertainty induced may be systematic. Nevertheless the uncertainty associated with these issues is important and needs to be addressed.

The impact pathway approach used for the externality analysis conducted here proceeds through a series of stages, each stage bringing in one additional parameter or component (e.g. data on stock at risk, a dose-response function, or valuation data) to which some degree of uncertainty can be linked. For statistical uncertainty one can attempt to assign probability distributions for each component of the analysis and calculate the overall uncertainty of the damage using statistical procedures. That is the approach recommended and adopted in this study (see below). In practice this is problematic because of the wide variety of possibly significant sources of error that are difficult to identify and analyse. In the following sections we present an estimate of the uncertainties of air pollution damages, to illustrate the approach.

For non-statistcal uncertainty it is more appropriate to indicate how the results depend on the choices that are made, and hence sensitivity analysis is more appropriate.

2. Analysis of Statistical Uncertainty

To determine the uncertainty of the damage costs, one needs to determine the component uncertainties at each step of impact pathway analysis and then combine them. For each parameter we have an estimate around which there is a range of possible alternative outcomes. In many cases the probability of any particular outcome can be described from the normal distribution with knowledge of the mean and standard deviation (σ) of the available data (Figure 1). The standard deviation is a measure of the variability of data: the zone defined by one standard deviation either side of the mean of a normally distributed variable will contain 68.26% of the distribution; the zone defined by the standard deviation multiplied by 1.96 contains 95% of the distribution etc.

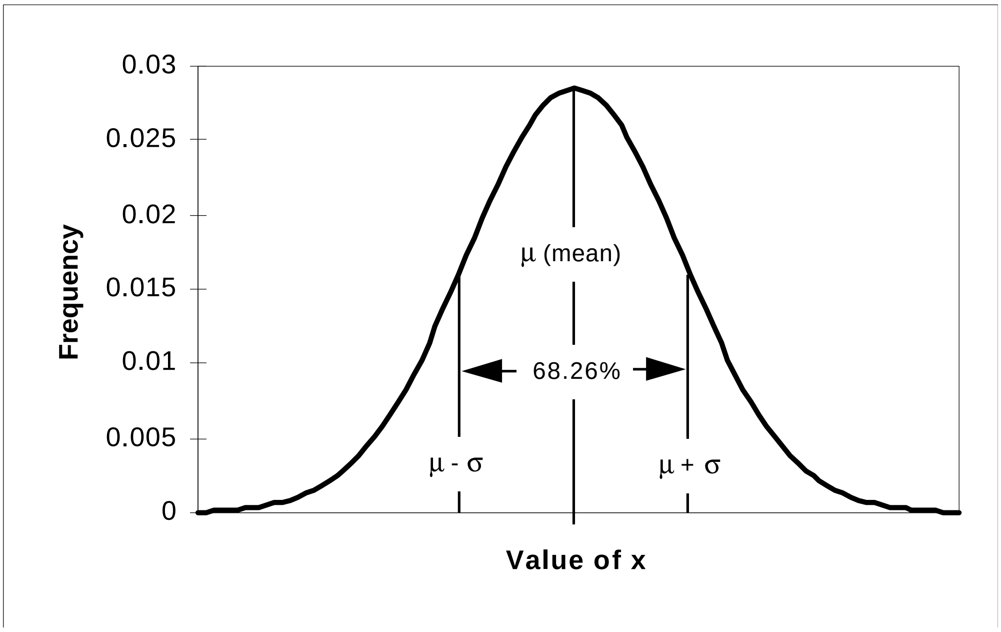

Figure 1. Illustration of the normal distribution.

The impact pathway analysis is typically multiplicative. For example, air pollution effects on health are calculated thus:

Damage = pollution concentration x population x exposure-response function x valuation

The distribution of outcomes from such a multiplicative analysis is typically lognormal, in other words the log of the variable is distributed normally. Plotted on a linear scale the lognormal distribution is skewed with the peak towards the left hand side (low values) and a tail to the right (high values) that may include extremely high outcomes, although with a low probability. By carrying out the log transformation the data become amenable to the statistical procedures that apply to the normal distribution.

This characteristic allows the use of multiplicative confidence intervals. Even though the complete characterisation of uncertainty requires an entire probability distribution rather than just a single number or interval, one can often assume that the distributions are approximately lognormal for multiplicative processes. In such cases the error distribution of the product approaches the lognormal distribution in the limit where the number of factors goes to infinity. In practice the approach to lognormality is quite close even when there are only a few factors, provided the distributions of these factors are themselves not too different from lognormal. Examples indicate that this is indeed a good assumption for the impact pathway analysis, and lognormality is a good approximation for the uncertainty analysis of the damage cost (Rabl, 1996).

To adopt this approach it is sufficient to specify just two numbers: the geometric mean (μ_g) and the geometric standard deviation (s_g). For the lognormal distribution, $\mu_g \cong$ median. By definition a variable x has a lognormal distribution if log(x) is normal. In the limit of small uncertainties, which are common in the physical sciences, s_g approaches 1 and the lognormal distribution approaches the normal. In field sciences, both biological and social, larger uncertainties are common, so that $s_g >> 1$.

With a normal distribution the confidence range with which a particular value can be predicted is determined by the mean (μ) and standard deviation (σ). Figure 1 illustrated the way in which confidence defined limits can be set around the mean using the standard deviation. With the lognormal distribution the confidence interval is predicted from the geometric mean (μ_g) and the geometric standard deviation (σ_g). Because of the properties of logarithms under addition, the relationship is additive for the logarithm of the variable, but multiplicative for the variable itself. The 68% confidence limits are then defined by the range μ_g/s_g to $\mu_g.s_g$ and the 95% confidence limits by the range μ_g/s_g^2 to $\mu g.s_g^2$.

As an illustrative example, Table 1, consider the acute mortality due to particulates. If the geometric standard deviations for each step are estimated to be

Emission	1.1;
Dispersion	2.5, and
Dose-response function	1.5,

one finds that the geometric standard deviation of the physical damage is $s_g = 2.7$, by using the formula:

$$[\log(s_{g,tot})]^2 = [\log(s_{g,1})]^2 + [\log(s_{g,2}]^2 + [\log(s_{g,3})]^2 \qquad 1.$$

for the combination of geometric standard deviations. If the median damage has been found to be $\mu_g = 2$ deaths/year, the one s_g interval is 2/2.7 = 0.74 to 2*2.7 = 5.4 deaths/year. This result provides an indication of the likely range of outcomes based on statistical uncertainties.

Table 1 Sample calculation of the geometric standard deviation for acute mortality due to air-borne particulates.

Stage	**Geometric standard deviation s_g**
Emission	1.1
Dispersion	2.5
Dose-response function	1.5
$s_{g,tot}$ for damage	2.7
Economic valuation	3.4
$s_{g,tot}$ for cost	4.9
Effects not taken into account	>1.0
Grand Total s_g	>4.9

In this indicative calculation, air pollution damages can be estimated to within about a factor of about five (68% confidence interval), excluding model, ethical and scenario uncertainties.

The actual values of the statistical uncertainties are considered in more detail in the next section for the case of air pollution effects linked to fossil fuel combustion, with the issues requiring additional sensitivity analysis in the following section.

3. Key Uncertainties

There are innumerable sources of uncertainty. Not all are equally important, and the relative importance of some is not immediately obvious. The approach taken here is to estimate the uncertainty of data we use at each step in the impact pathway, so these can be combined with the approach illustrated in the previous section. In most cases, uncertainties are fairly similar across a range of pathways for the same pathway step. We therefore attempt only to estimate a typical value for each pathway stage.

3.1 Emissions

This step generally has the lowest uncertainty of all those considered at least for the macropollutants (SO_2, NO_x, CO_2, etc). Many emissions are measured, either continuously, or by sample tests for smaller emitters. National data is probably good to an accuracy of about 10%. The value of σ for this step is therefore estimated to be 1.1.

3.2 Dispersion/Transformation of Pollutants

Uncertainty estimates for pollution calculations are difficult to determine, but data in the literature suggest that for annual average concentrations (which are of most interest here) the uncertainty may be a multiplicative interval of about a factor of two to three. The value of σ for this step is therefore estimated to be 2.5.

3.3 Exposure-response Functions for Air Pollution Effects

3.3.1 Health Impacts

For the key impacts on mortality, the dose-response function are based on major epidemiological studies. Considerable doubts remain about the interpretation of these studies in terms of the pollutants responsible and the existence of chronic impacts. These impact on the choice of study used and how it is interpreted - issues which are considered further in the section. Well conducted epidemiological studies report detailed statistical analyses. From these a typical value of σ for this step is therefore to be 1.5.

3.3.2 Impacts on Crops

Ozone damage costs are the only agriculture effects quantified in this study. There are uncertainties relating to the use of results based on only a few crops exposed under laboratory conditions for describing effects across the whole of Europe. These are considered in the next section. The results of the studies themselves have rather low uncertainty. The value of σ for this step is therefore estimated to be 1.5??.

3.3.3 Impacts on Buildings and Materials

????The value of σ for this step is therefore estimated to be ???

3.4 Economic Valuation

3.4.1 Health Effects

Economic valuation studies of mortality reported in the literature have been subjected to meta-analysis (Ives *et al*). The range of values is large. From consideration of published data the value of σ for this step is therefore estimated to be 3.4.

3.4.2 Impacts on Crops

?

3.4.3 Impacts on Buildings and Materials

?

3.5 Quantified Uncertainties

The parameters identified above have been used to derive estimates of σ_g (the geometric standard deviation). These have been placed in three bands;
?A
B
C

These bands are reported impact by impact elsewhere within this report. Given that the parameters needed for this analysis have all been quantified in this section, it is reasonable to ask why the final result is given as a band, rather than the number calculated for each type of impact. The reason is that the data given in this section are themselves uncertain. To give a single figure would imply greater confidence in the characterisation of uncertainty than really exists.

It is to be remembered that the 95% confidence interval is calculated by dividing/multiplying μ by σ_g^2. The overall ranges represented by the confidence bands are therefore larger than they might at first appear; band C actually covers four orders of magnitude.

4. Key Sensitivities

There are important issues in model choice at almost all stages of the analysis. Models have different credibility depending upon the quality of analysis which underpins them and the extent to which they have been validated. In addition, application of even the best models generates some additional concerns, relating to their use over a range of times and places and for purposes different from those intended by their authors.

For impacts which extend far into the future, the nature of the underlying world on which the impacts are imposed is fundamentally undetermined. Assumptions are necessary, but different scenarios for the relevant background conditions (environmental and social) can generate different results.

In addition, some issues, notably discounting, are controversial because they have substantial moral and ethical implications. It is important for decision making that these are integrated into the analysis in a transparent manner. They should therefore be treated explicitly as sensitivities and not simply be assumed to take the values the analysts prefer.

The approach used here is to identify sensitivities which are potentially important in the sense that they both:

- materially affect the magnitude of the damages calculated, and
- are variations on the baseline assumptions which are not unreasonable to experts in the field.

The following sections aim to identify those issues which meet these criteria. These are then used in the sensitivity analyses.

4.1 Emissions

For the major air pollutants of concern to externalities analysis, emissions are well known (to within a few percent). For micropollutants this is much less the case. For dioxins there is doubt about not just the total

4.2 Dispersion/Transformation of Pollutants

For most air pollutants the range over which they are transported is of the order of hundreds to thousands of km. In addition to issues related to the physical movement of pollutants through the air it is also necessary to consider the chemical transformations that occur. SO_2 is gradually oxidised to sulphate, NO to NO_2 and then nitrate. It is these secondary particulates (sulphates and nitrates) that are most closely linked with the biggest health effects. Also, NO_x is involved in the formation of ozone, which is also involved in the chemistry leading to formation of sulphate and nitrate aerosol. The model used has been extensively tested against empirical data in Europe. At least for rural measurements, agreement is reasonable and we certainly have no reason to expect that there are large systematic errors.

Some pollutants will pass outside the modelled area and cause damages there. However, this is believe to be a minor effect compared to the damages of european emissions in Europe and is therefore neglected.

Consideration must also be given to the effect of emissions location on the damage and the extent to which the modelling procedures used address this sensitivity. Previous studies by ARMINES have shown that the health damage from air pollution varies by about a factor of ten for a hypothetical plant located in Paris, compared with one at a rural site on the Atlantic Coast. The implication is that, even within a single grid cell, there can be significant variations due to population density, which the coarse scale of the EMEP model may be unable to capture. This is particularly true for emissions at ground level in urban areas, where the expected damages may be many times that predicted by regional scale models (Eyre *et al*, 1997).

The situation for ozone is even more difficult, because in many cases NO_x concentrations are sufficiently high in urban areas that ozone formation is VOC limited. NO_x emissions (primarily NO) then reduce ozone concentrations. The reaction which does this is incorporated in the EMEP chemistry scheme, but the coarse grid scale prevents its importance in urban locations being modelled accurately.

Key sensitivities to be investigated are therefore the effect of different assumptions about the model validity for urban populations, plus??

4.3 Dose-response Functions for Health Impacts of Air Pollution

A consensus has been emerging among public health experts that air pollution, even at current ambient levels, causes a variety of significant health problems, especially respiratory diseases and increased mortality (Lipfert, 1994; Dockery and Pope, 1994). There is less certainty about specific causes, but most recent studies have identified fine particulates as a prime culprit and ozone has also been implicated directly. There may also be significant direct health impacts of SO_2, but for direct impacts of NO_x the evidence is not convincing.

In the ExternE Program, for example, the working hypothesis has been to use the health dose-response functions for particulates and ozone. Effects of NO_2 and SO_2 are assumed to arise indirectly from the particulate nature of nitrate and sulphate aerosols, and they are calculated by applying the particulate dose-response functions to the aerosol concentrations. With this assumption the impacts of nitrate aerosols become very large, but this is very uncertain because there is no direct evidence for health impacts of nitrate aerosols (in contrast to sulphate aerosols). It is quite possible that the non-ozone health effects of NO_2 are very small or even zero.

Most dose-response functions for health effects from air pollution are determined by short term correlations (time lags of a few days at most) and thus they measure only acute effects. In view of the weight of the epidemiological evidence, the numbers for acute mortality due to the release of particulates are quite firm and they can be considered lower bounds. But the full impact includes chronic effects which can be much larger, albeit notoriously difficult to measure. Two recent studies suggest that the true mortality maybe several times larger than the acute mortality because of chronic effects (Dockery *et al*, 1993; Pope *et al*, 1995).

In this project the dose-response functions have been assumed linear, in view of the lack of evidence for thresholds at current ambient concentrations.

Key sensitivities to be investigated are therefore the effect of thresholds, different assumptions about chronic health impacts, and different assessments of the impact of aerosol in comparison to total particulate. Plus??

4.4 Impacts on Crops

Dose-response functions are reasonably well established for certain economically important crops such as wheat and rye. Particulates do not have a significant impact on crops, and NO_x is a fertiliser. Sulphur dioxide may be beneficial to crops at low doses and harmful at high doses. Its damage from this pathway have been estimated and found to be small compared to health costs. Ozone damages will depend on the applicability of measured dose-response function to crops in the field in water limited conditions and the extent to which damage to pasture grass in reflected in reduced production of meat and milk. Key sensitivities to be investigated are therefore the reductions on ozone sensitivity below those implied by laboratory measurements.

4.5 Impacts on Buildings and Materials

Dose-response functions are more or less established for the physical impacts of SOx, and to some extent NO_x, PM, and O_3, on certain important materials (e.g. galvanised steel, stone, plaster, paints). Their application in damage cost estimates is, however, problematic because of lack of information on inventories of exposed materials and about the economic valuation. One finds a wide range of damage cost estimates, from very small (Rabl *et al*, 1996) to significant (EC, 1995), but in any case less than the health costs. Key sensitivities to be investigated are therefore ??

4.6 Economic Valuation

The value of statistical life (VSL) is one of the key parameters in this study because, at the assumed value of 3 MECU, mortality impacts dominate all else. There is, however, a question mark over the applicability of VSL to pollution related deaths. By using a single VSL to value death, one does not take into account the age at death or the number of years of life lost. No one has any precise idea about length of life lost, and the current practice of using the full VSL with the acute mortality dose-response functions may not be appropriate if the shortening of life is only a few weeks or months. The concept of years-of-life-lost (YOLL) has been presented as an alternative but has not been applied in most of the recent fuel cycle studies, in particular not in ExternE. the sensitivity of damages to different assumptions about valuation of life are therefore critical.

Most impacts of air pollution are fairly immediate and discounting is not significant. The main exceptions are chronic health impacts and cancers. For chronic health impacts, in particular the so-called chronic mortality from particulates, the appropriate discounting period is not known with any precision - perhaps a period on the order of ten years is appropriate. At a 3%

discount rates, costs are reduced by a factor of 1.3, not a major effect in view of other uncertainties. For cancers, the latency period is likely to be longer and not all cancers are fatal. Hence our cancer damage cost estimates are probably too high. Key sensitivities to be investigated are therefore the effect of discount rate on cancer deaths??

5. Conclusions

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