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Edited by Lotte Schleisner and Per Sieverts Nielsen

Risø National Laboratory, Roskilde, Denmark
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ABSTRACT

The objective of the ExternE National Implementation project has been to establish a comprehensive and comparable set of data on externalities of power generation for all EU member states and Norway. The tasks include the application of the ExternE methodology to the most important fuel cycles for each country as well as to update the already existing results, to aggregate these site- and technology-specific results to more general figures.

The current report covers the detailed information concerning the ExternE methodology. Importance is attached to the computer system used in the project and the assessment of air pollution effects on health, materials and ecological effects. Also the assessment of global warming damages are described. Finally the report covers the detailed information concerning the national implementation for Denmark for an offshore wind farm and a wind farm on land, a decentralised CHP plant based on natural gas and a decentralised CHP plant based on biogas.

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CONTRIBUTORS

This report states the background information for the ExternE Implementation Project.

The authors responsible of the different appendices are as follows:

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Appendix II: Health effects	F. Hurley, P. Donnan, IOM, Edinburgh, Scotland
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Appendix IX: Definition of the biogas fuel cycle	Nielsen, P.S. Technical University of Denmark Denmark
Appendix X: Definition of the wind fuel cycles	Schleisner, L. Risø National Laboratory, Denmark

Appendices I-VII are general appendices in all the respective country reports, describing the methodology and background data used in all countries for the National Implementation Project. Appendices VIII-X are appendices specific for the Danish implementation.

Mike Holland has compiled Appendices 2, 3, 4, 5 and 6.

CONTENTS

I. THE ECOSENSE MODEL

I.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.

I.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, EcoSense does not cover this impact category 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.

I.3. The EcoSense Modules

Figure I.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.

I.3.1. The EcoSense database

I.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.

I.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 system, which defines equal-area projection gridcells of 10 000 km^2 and 100 km^2 (Bonnefous a. 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.

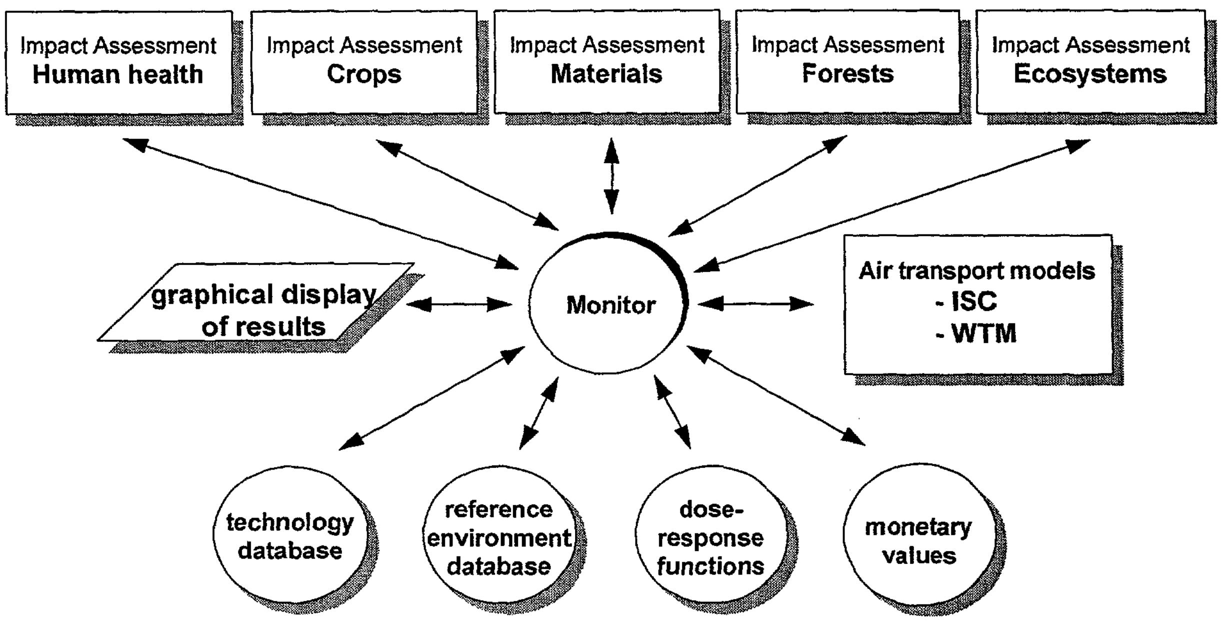

Figure I.1 Structure of the EcoSense model

I.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.

I.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.

I.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, and 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.

I.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 automised analysis including a range of prespecified impact categories.

I.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.

I.4. The air quality models integrated in EcoSense

I.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:

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

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

- The mixing height is terrain following.

Mean terrain heights for each grid cell are provided by the reference environment database. However, it should be mentioned that the application of a Gaussian plume model to regions with complex topography is problematic, so that in such cases better adapted models should be used if possible.

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.

I.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 I.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).

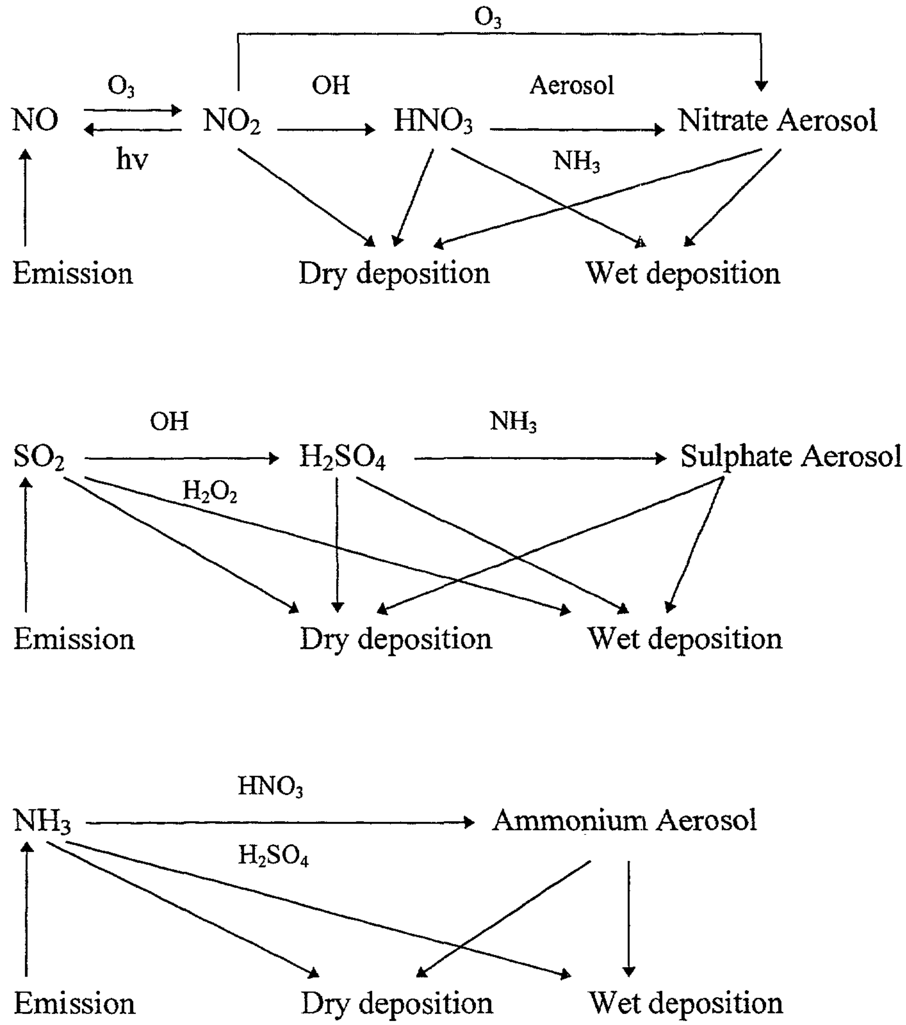

Figure I.2 Chemical Scheme in WTM, adopted from Derwent et al. (1993)

I.5. References

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II. HEALTH EFFECTS

II.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, followed by a review of valuation issues for health effects. 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, 1995e). It has to be noted that, since the results of ExternE 1995 (European Commission, 1995a-f) were published, a lot of new information has become available, changing the quantification and valuation of some health impacts significantly.

II.2. Non-Carcinogenic Effects of Air Pollutants

II.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, 1995c, 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.

II.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 Table II.1 and Table II.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 II.1 contains the 'core' set of exposure-response functions used in ExternE. Table II.2 contains functions recommended only for use in sensitivity analysis.

Table II.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},	0.163	B
			Nitrates,	0.163	B?
			$PM_{2.5}$,	0.272	B
			Sulphates	0.272	B
	Cough	Dusseldorp *et al*, 1995	PM_{10},	0.168	A
			Nitrates,	0.168	A?
			$PM_{2.5}$,	0.280	A
			Sulphates	0.280	A
	Lower respiratory symptoms (wheeze)	Dusseldorp *et al*, 1995	PM_{10},	0.061	A
			Nitrates,	0.061	A?
			$PM_{2.5}$,	0.101	A
			Sulphates	0.101	A
children	Bronchodilator usage	Roemer *et al*, 1993	PM_{10},	0.078	B
			Nitrates,	0.078	B?
			$PM_{2.5}$,	0.129	B
			Sulphates	0.129	B
	Cough	Pope and Dockery, 1992	PM_{10},	0.133	A
			Nitrates,	0.133	A?
			$PM_{2.5}$,	0.223	A
			Sulphates	0.223	A
	Lower respiratory symptoms (wheeze)	Roemer *et al*, 1993	PM_{10},	0.103	A
			Nitrates,	0.103	A?
			$PM_{2.5}$,	0.172	A
			Sulphates	0.172	A
all	Asthma attacks (AA)	Whittemore and Korn, 1980	O_3	4.29E-3	B?
ELDERLY 65+ (14% of population)					
	Congestive heart failure	Schwartz and Morris, 1995	PM_{10},	1.85E-5	B
			Nitrates,	1.85E-5	B?
			$PM_{2.5}$,	3.09E-5	B
			Sulphates,	3.09E-5	B
			CO	5.55E-7	B
CHILDREN (20% of population)					
	Chronic bronchitis	Dockery *et al*, 1989	PM_{10},	1.61E-3	B
			Nitrates,	1.61E-3	B?
			$PM_{2.5}$,	2.69E-3	B
			Sulphates	2.69E-3	B

Receptor	Impact Category	Reference	Pollutant	f_{er} [1]	Uncertainty rating
	Chronic cough	Dockery *et al*, 1989	PM_{10},	2.07E-3	B
			Nitrates,	2.07E-3	B?
			$PM_{2.5}$,	3.46E-3	B
			Sulphates	3.46E-3	B
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

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

[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 II.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}, Nitrates, $PM_{2.5}$, Sulphates CO	1.75E-5 1.75E-5 2.92E-5 2.92E-5 4.17E-7	B B? B B 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} Sulphates, $PM_{2.5}$	7.20E-6 1.20E-5	B? B?
	ERV for asthma	Schwartz, 1993 and Bates *et al*, 1990 Cody *et al*, 1992 and Bates *et al*, 1990	Nitrates, PM_{10} Sulphates, $PM_{2.5}$ O_3	6.45E-6 1.08E-5 1.32E-5	B? B? B?
	ERV for croup in pre school children	Schwartz *et al*, 1991	Nitrates, PM_{10} Sulphates, $PM_{2.5}$	2.91E-5 4.86E-5	B? B?
	Cancer risk estimates	Pilkington and Hurley, 1997	Formaldehyde	1.43E-7	B?
	Acute Mortality (AM)	Touloumi *et al*, 1994 Sunyer *et al*, 1996, Anderson *et al*, 1996	CO NO_2	0.0015% 0.034%	B? B?

[1] Sources: [EC, 1995b] 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 ExternE 1995, 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.

II.3. Carcinogenic Effects of Radionuclide Emissions

II.3.1. Introduction

A brief explanation of the terminology specific to the nuclear fuel cycle assessment is presented in Box 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. The details given relate specifically to the French implementation of the nuclear fuel cycle (European Commission, 1995e). For the implementation in the present phase of the study, a more simplified approach has been adopted by some teams that extrapolates from the French results.

Box 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.

For assessment of 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.

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.

II.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 or discussed in a time and space matrix. This form of presentation of results makes 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.

II.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 II.1.

II.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).

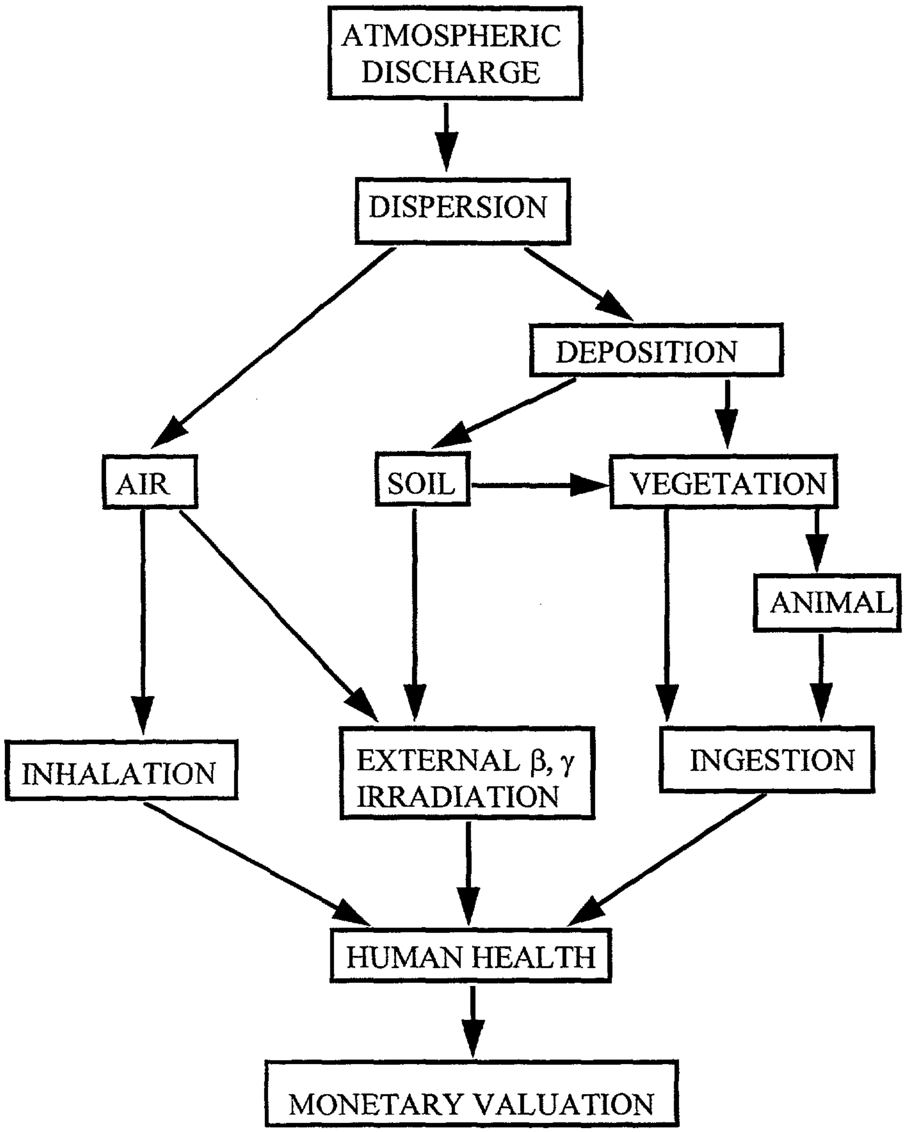

Figure II.1 Impact pathway for an atmospheric release of radionuclides into the terrestrial environment

II.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 arises from 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 should be sufficient.

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 II.3.

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 II.4) 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.

II.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 II.5.

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.

Table II.3 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 II.4 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

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, 1995e) in a way that allows the reader to apply different factors if desired.

The major damages from the nuclear fuel cycle result from a large number of people being affected by very low doses. Therefore, the linearity of the dose response function is a fundamental assumption. However, there is no incontestable scientific evidence today to support the threshold nor Hormesis effect. Therefore, the ICRP recommends the conservative approach of assuming a linear dose-response function which continues to zero dose.

Table II.5 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

II.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 have 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.

II.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 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.

II.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.

II.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 European population consumes the edible portion of the food harvested in the northern European waters 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.

II.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 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.

II.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:

$$\mathbf{Risk} = \sum P_i \cdot C_i$$

II.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.

II.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.

II.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.

II.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).

II.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 II.6.

Table II.6 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.

II.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.

II.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 II.7. Safety factors reflect the uncertainty involved in extrapolating data between species and also the perceived severity of the effect.

Table II.7 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).

II.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 II.6.

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 II.8 summarises the results for the plant emitting the highest levels of dioxins considered by HMIP.

Table II.8 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 II.8 that even in the worst case considered, of an urban HMEI living near the largest plant emission considered by the study, the total intake does not approach the

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

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

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

EF: exposure frequency (hours/year)

ED: exposure duration (years)

BW: body weight (kg)

AT: averaging time

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

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

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 II.9.

Table II.9 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 Inhalation Fraction of 0.02 for Dioxins (relative exposure via inhalation = 2%).

No-threshold assumption:

Unit risk factor from

LAI 1.4 per $\mu g/m^3$

leading to the following ERF implemented in EcoSense:

(1) No. of additional cancers = Δ Concentration [$\mu g/m^3$] * 1.4 * Population /70

Threshold assumption:

WHO 'tolerable daily intake': 10 pg/($kg_{BW} \cdot d$)

Using an Inhalation Fraction of 0.02 (relative exposure via inhalation = 2 %), the air concentration equivalent to the threshold dose is 5.4 E-7 $\mu g/m^3$.

Background:

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

II.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).

II.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 exceedence of the WHO guidelines does happen, so effects cannot be ruled out.

II.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.

II.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 Inhalation Fraction of 0.004 (relative exposure via inhalation = 0.4 %), the air concentration equivalent to the threshold dose is 3.3 ng/m^3

Background:

WHO-Air Quality Guidelines 1987:

rural areas:	1-10 ng/m^3
urban areas:	< 1 µg/m^3

France (see Rabl, 1996): 1 - 4 ng/m^3

LAI (Germany)

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

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

II.4.2.4 Chromium

Cancer

Unit risk factor from

WHO (1987) 0.04 per $\mu g/m^3$

Non-carcinogenic effects

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

II.4.2.5 Nickel

Cancer

Unit risk factors from

WHO (1987) 0.0004 per $\mu g/m^3$

US-EPA (1996) 0.004 per $\mu g/m^3$

Non-carcinogenic effects

Threshold:

ATSDR (1996) 0.02 $mg/(kg_{BW} \cdot d)$

Using the equations derived above, and an Inhalation Fraction of 0.003 (relative exposure via inhalation = 0.3 %), the air concentration equivalent to the threshold dose is 0.16 $\mu g/m^3$ 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.

II.5. Occupational Health Issues (Disease and Accidents)

II.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 essential 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 occurred 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 artificially 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 safety 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.

The ExternE methodology aims to quantify all occupational health impacts, including those outside Europe, e.g. linked to the mining and treatment of imported fuels. Within the present phase of the study an assessment of damages outside Europe became 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.

II.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.

II.7. Valuation

II.7.1. Introduction

Valuation of health effects can be broken down into the following categories;

- Mortality linked to short term (acute) exposure to air pollution
- Mortality linked to long term (chronic) exposure to non-carcinogenic air pollutants
- Mortality from exposure to hazardous materials in the workplace
- Mortality from cancer
- Morbidity from short term (acute) exposure to air pollution
- Morbidity from long term (chronic) exposure to air pollution
- Morbidity from exposure to hazardous materials in the workplace
- Mortality from workplace accidents
- Injury from workplace accidents
- Mortality from fuel chain accidents that affect the public
- Injury from fuel chain accidents that affect the public

This section briefly reviews the data used and some of the issues linked to health valuation.

II.7.2. Mortality

The value of statistical life (VSL), essentially a measure of WTP for reducing the risk of premature death, is an important parameter for all fuel chains. A major review of studies from Europe and the US , covering three valuation methods (wage risk, contingent valuation and consumer market surveys) is described in earlier work conducted by the ExternE programme

(European Commission, 1995b). The value derived for the VSL was 2.6 MECU. This value has been adjusted to 3.1 MECU, to bring it into line with January 1995 prices, as has been done for all valuations.

However, in earlier phases of the project a number of questions were raised regarding the use of the VSL 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. Was it 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 VSL to cases where air pollution is only one factor of perhaps several that determines the time of death, with air pollution playing perhaps only a minor role in the timing of mortality?. In view of this the project team explored valuation on the basis of life years lost. For quantification of the value of a life year (YOLL) it was necessary to adapt the estimate of the VSL. This is not ideal by any means (to derive a robust estimate primary research is required), but it does provide a first estimate for the YOLL.

A valid criticism of the YOLL 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 VSL 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 VSL 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 VSL. 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 VSL 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 for which the VSL applies. The view of the project team is that the VSL 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 (YOLL) lost approach is recommended. However, in view of the continuing debate in this area among experienced and respected practitioners, and continuing and genuine uncertainty, the VSL has been retained for sensitivity analysis.

The YOLL 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 impacts caused by fine particles: one for acute mortality and for chronic mortality (Table II.10). Both sets vary with discount rate.

Table II.10 Estimated YOLL for acute and chronic effects of air pollution at different discount rates, and best estimate of the VSL.

Type of effect/discount rate	YOLL (1995ECU)
Acute effects on mortality	
0%	73,500
3%	116,250
10%	234,000
Chronic effects on mortality	
0%	98,000
3%	84,330
10%	60,340
Estimated value of statistical life	3,100,000

II.7.3. Morbidity

Updated values for morbidity effects are given in Table II.11. Compared to the earlier report (European Commission, 1995b) 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 II.11 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.

Endpoint	New Value	Estimation Method and Comments
Malignant Neoplasms	450,000	Valued as non-fatal cancer.
Chronic Case of Asthma	105,000	Based on treating chronic asthma as new cases of chronic bronchitis.
Cases of change in prevalence of bronchitis in children	225	Treated as cases of acute bronchitis.
Cases of change in prevalence of cough in children	225	As above.

II.7.4. Injuries

The following data (Table II.12) have been provided by Markandya (European Commission, 1998).

Table II.12 Valuation data for injuries.

Endpoint	Value (ECU, 1995)	Estimation method and comments
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 costs 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.

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III. AIR POLLUTION EFFECTS ON MATERIALS

III.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 'utilitarian' 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).

III.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 for which 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 have been used. 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

III.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).

III.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 III.1 1.

Table III.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.

III.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):

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

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):

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

ICP - sheltered limestone (4 years):

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

ICP - sheltered sandstone (4 years):

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

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.

III.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.

III.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.

III.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:

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

Haynie - silicate paint:

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

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.

III.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 typically coated with paint when not galvanised (see section 4.5.1 of this Appendix). The stock of steel in our inventories has therefore been transferred to the paint stock at risk.

III.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.

III.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):

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

III.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 III.2. The figures given in Table III.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 III.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

III.6. Estimation of economic damage (repair costs)

The valuation of impacts should ideally be made in terms of the willingness to pay to avoid 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 III.3 summarises the damage costs used in this analysis in 1995ECU.

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

Material	ECU/m^2
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.

III.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.

III.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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IV. ANALYSIS OF ECOLOGICAL EFFECTS

IV.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. It is based on an earlier review under the ExternE Project (European Commission, 1995a) which has been updated by Jones *et al* (1997, for inclusion in the updated ExternE Methodology Report, European Commission, 1998a).

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 hoped that further progress will be made in the near future.

There 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, 1998b).

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

IV.2. Air Pollution Effects on Crops

IV.2.1. SO_2 Effects

A limited number of exposure-response functions dealing with direct effects of SO_2 on crops are available. 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. However, it has been demonstrated in a large number of experiments that low levels of SO_2 are capable of stimulating growth; therefore 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 illustration of the extrapolation procedure is shown in Figure IV.1.

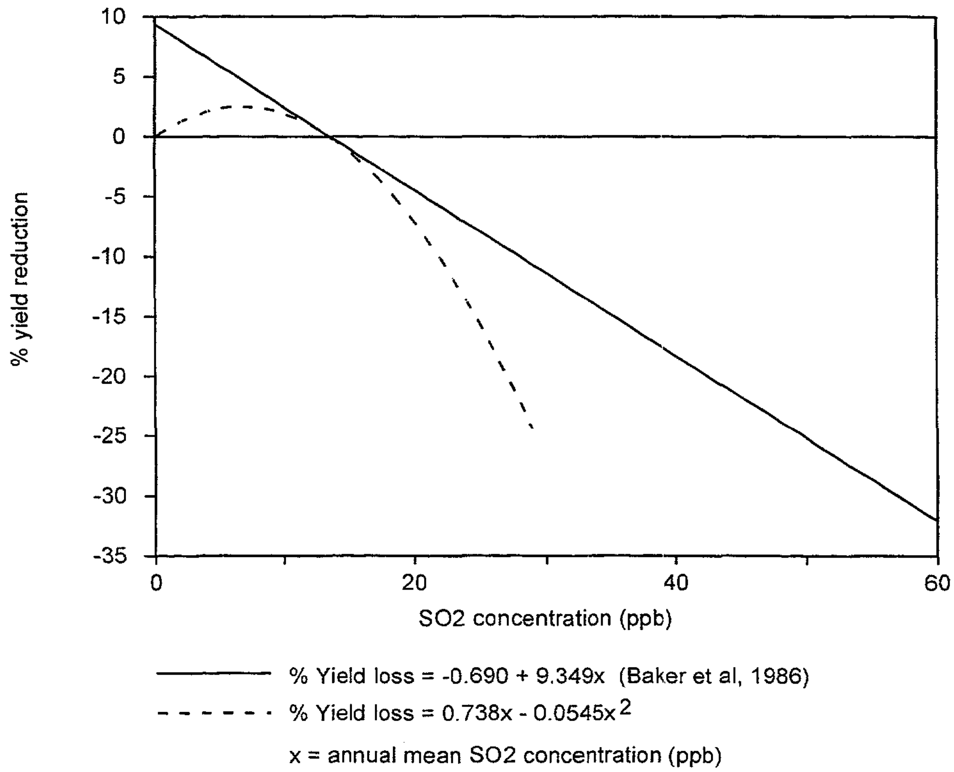

Figure IV.1 Extrapolation of exposure-response functions below the lowest exposure level used experimentally.

Baker *et al* (1986) reported that weather conditions varied greatly between years in their experiment; *'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, respectively; 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 $\mu g\ m^{-3}$ (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 $\mu g\ m^{-3}$. Daily means of NO_2 and NO concentrations were generally lower than 10 $\mu g\ m^{-3}$. 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 (McGrath and Withers, 1996). Data for barley were taken from this paper and used to calculate the following relationship (SO_2 in $\mu g\ m^{-3}$):

$$y = 10.92 - 0.31(SO_2) \qquad (3)$$

$r^2 = 0.73$, $p < 0.01$, 10 data points for barley only

($y = 10.92 - 0.89(SO_2)$, SO_2 in ppb)

The background mean SO_2 concentrations that provided the control levels in this study were low (7 - 9 $\mu g\ m^{-3}$, about 3 ppb). It is considered that function 3, unlike function 1, may thus be applied directly without the need to consider how best to extrapolate back to 0 ppb SO_2. The two functions (2a/2b and 3) could be said to operate under alternative circumstances, one where soil sulphur levels are too low for optimal growth, and the other where they are sufficient.

Function 2 was recommended to derive best estimates for changes in crop yield for wheat, barley, potato, sugar beet, rye and oats. For sensitivity analysis function 1 for all crops and function 3 for barley have been used. 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, though of course the importance of such interactions will vary extensively from site to site.

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 detoxify any SO_2 absorbed.

IV.2.2. O_3 Effects

Complete details of the assessment of ozone damages under the ExternE Project are given elsewhere (European Commission, 1998a). 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:

$$y_r = a \cdot e^{-(x/s)^c} \tag{4}$$

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.

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

$$\text{reduction in yield per ppb} = \frac{1}{y}\ dy/dConc \qquad (5)$$

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*, 1993). However, only 3 crops were covered, spring wheat, oats and barley, the last 2 of which 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.

Table IV.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

A general reluctance to use exposure-response functions for ozone effects in Europe is noted, largely as a consequence of the uncertainties introduced through interactions, particularly with water stress. Peak ozone episodes tend to occur with hot spells when plants are most likely to be water stressed. Stomatal conductance under such conditions is reduced to prevent water loss, which of course also reduces the uptake rate for ozone. Offsetting this, a substantial amount of land is irrigated in southern European countries where the effect is likely to be greatest (Eurostat, 1995). This will tend to be concentrated on higher value crops. In the context of this study we believe that it is preferable to quantify damages than to ignore them, provided that uncertainties are noted.

The following function was derived for sensitive crops;

$$Y_{rel} = 1 + 0.0008 \cdot x_8 - 0.000075 \cdot x_8^{\ 2} \qquad (6)$$

Where Y_{rel} = relative yield

x_8 = average daily peak 8 hour concentration.

The various functions shown in this Appendix were used to generate an average function for crop loss (Table IV.2) which was applied to crops not covered by the functions in Table IV.1.

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. Based on the these model results and the listed ozone crop functions an ozone crop damage factor of 490 ECU per tonne NO_x emitted in Europe has been derived.

Table IV.2. Average and standard deviation of yield reduction for species in Figure IV.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
$\frac{1}{y}$ dy/dConc from d-r function	-0.0058	0.0033
(y - 1)/Conc straight line	-0.0025	0.0014

IV.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 analysis calculates the amount of lime required to balance acid inputs on agricultural soils across Europe. Analysis of liming needs should of course be restricted to non-calcareous soils. However, the percentage of the agricultural area on non-calcareous soils has not been available Europe-wide. Thus, the quantified additional lime required is an over-estimate giving an upper limit to the actual costs.

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);

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

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. Further details were given in European Commission (1995a).

IV.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).

IV.3. Modelling Air Pollution Damage to Forests

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. The 1995 ExternE report paid particular attention to the studies by Sverdrup and Warfvinge (1993) and Kuylenstierna and Chadwick (1994), and functions developed by FBWL (1989) and Kley *et al* (1990). However, although we regard these approaches as worthy of further consideration, the results that they provide are too uncertain for application at the present time in support of policy development.

Kroth *et al* (1989) assessed the silvicultural measures which forest managers apply to counteract forest damages, and associated costs for Germany. Using the specific costs Kroth *et al* calculated totals for the whole of West Germany. Taking into account only those measures, which have been approved by experts to have mitigating potential and which are separable from normal operation, total costs for West Germany of 41.2 to 112.9 MECU/year have been quantified for a five year period.

The total figure can be divided by the total area of damaged forest according to the forest damage inventory to provide an estimate of cost per hectare over a five year period. Multiplying this by the incremental increase in forest damage area 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.

IV.4. Assessment of Eutrophication Effects on Natural Ecosystems

In addition to acidification, inputs of nitrogen may cause an eutrophication of ecosystems. Too high nitrogen inputs displace other important nutrients or impair their take-up (Matzner and Murach, 1995). This causes nutrient imbalances and deficiency symptoms. When the deposited nitrogen is not completely used for primary production, the excess nitrogen can be inactively accumulated in the system, washed out or emitted again as nitrous oxide (N_2O).

Furthermore, the competition between different populations of organisms is influenced, at the expense of species which have evolved to dominate in nutrient poor soils (Nilsson and Grennfelt, 1988; Breemen and Dijk, 1988; Heil and Diemont, 1983). Accordingly, the *critical load for nitrogen nutrient effects* is defined as

> "*a quantitative estimate of an exposure to deposition of nitrogen as NH_x and/or NO_x below which empirically detectable changes in ecosystem structure and function do not occur according to present knowledge*" (Nilsson and Grennfelt, 1988).

The UN-ECE has set critical loads of nutrient nitrogen for natural and semi-natural ecosystems (Table 3). The Institute of Terrestrial Ecology in Grange-over-Sands, UK, together with the Stockholm Environment Institute in York, UK, have produced critical load maps for nutrient nitrogen (eutrophication) for semi-natural ecosystems on the EUROGRID 100x100 km^2 grid by combining the critical loads of the UN-ECE with the a European land cover map.

Table IV.3 Areas and critical loads of nutrient nitrogen for natural and semi-natural ecosystems

Ecosystem	Ecosystem area [km^2]	Critical load of nutrient nitrogen [kg/ha/year]
Acid and neutral, dry and wet unimproved grass	564510	20-30
Alkaline dry and wet unimproved grass	226067	15-35
Alpine meadows	58548	5-15
Tundra/rock/ice	228218	5-15
Mediterranean scrub	82807	15
Peat bog	50601	5-10
Swamp marsh	23551	20-35
Dwarf birch Scots pine Spruce and/or fir Pine/spruce with oak/birch Pine/spruce with birch Maritime pine Stone pine Aleppo pine	1038997	10-15 (nutrient imbalance) 10-25 (nitrogen saturation) 7-20 (ground flora changes)
Beech Various oaks Cork oak Holm oak	525642	15-20 (nutrient imbalance) 10-20 (ground flora changes)

Source: UN-ECE (1996), Howard (1997)

One of the management rules for sustainability as defined by Pearce and Turner (1990) requires that the assimilative capacity of ecosystems should not be jeopardised. The critical level/load concept of the UN-ECE is a good basis to derive sustainability indicators with respect to this management rule.

Two types of indicators are available. First, the exceedence area (the area in which the respective critical load is exceeded). It has to be pointed out that the exceedence area difference does not necessarily equate with the difference in damage between the scenarios. Damages do not necessarily occur the moment the critical loads are exceeded nor is the impact necessarily proportional to the height of the exceedence. Consequently, the difference in exceedence area between scenarios could be large, but the damage difference might still be small. Conversely, the exceedence area difference could be zero, while the difference in damage is very large. Overall, therefore, the size of the exceedence area is only an indicator of the possible damage.

When the emissions of one facility are analysed the exceedence area difference between the background and the new scenario always is zero. The pollutant level increments due to emissions of one facility are of a much lower order of magnitude than the critical loads. It should also be kept in mind that there are many uncertainties attached to the setting of the critical loads that are higher than the pollutant level increments due to one facility. In essence the result for a single plant is meaningless. The *sensitivity limit* i.e. the minimum emission difference between two scenarios in order that the additional exceedence area is not zero, is different for each critical load map. *Inter alia* it depends on the number of critical load classes.

The second indicator type is based on the assumption that the higher the exceedence height in an area the larger the potential effect. The indicator takes the exceedence height into account by weighting the exceedence area with it:

$$A_{Exc,weighted} = \sum_{ij} \begin{cases} A_{\mathrm{Ecos},ij} \cdot \dfrac{C_{ij} - L}{L} & C_{ij} > L \\ 0 & C_{ij} \le L \end{cases} \qquad (7)$$

Where

ij	Index of EUROGRID grid cell
$A_{Ecos,ij}$	Area of ecosystem in grid cell ij
C_{ij}	Pollutant concentration or deposition for scenario under analysis in grid cell ij
L	Critical load

The exceedence height is normalised by the critical load with the effect that the more sensitive an ecosystem is (which is equivalent to a low critical load), the more the exceedence is valued. The indicator is called relative exceedence weighted exceedence area or potential impact weighted exceedence area.

An advantage of this indicator type is that the misinterpretation of no difference in exceedence area between two scenarios as no impact difference is avoided. Even if the critical load is already exceeded for the scenario with the lower emissions, the indicator difference is not zero but reflects the difference in pollutant levels. Therefore, the indicator also yields reasonable results when the difference in emissions between the two scenarios is small, as it is the case when a single power plant is analysed (no sensitivity limit as for the first indicator).

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V. ASSESSMENT OF GLOBAL WARMING DAMAGES

V.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 are included and excluded by the two models are provided by European Commission (1998).

V.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 feels so strongly about this that reference should not be made to the ExternE Project results unless reference is also made to the uncertainties inherent in any analysis and our attempts to address them. Reliance on any single number in a policy-related context will provide answers that are considerably less robust than results based on the range, although this, in itself, is uncertain.

V.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.

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

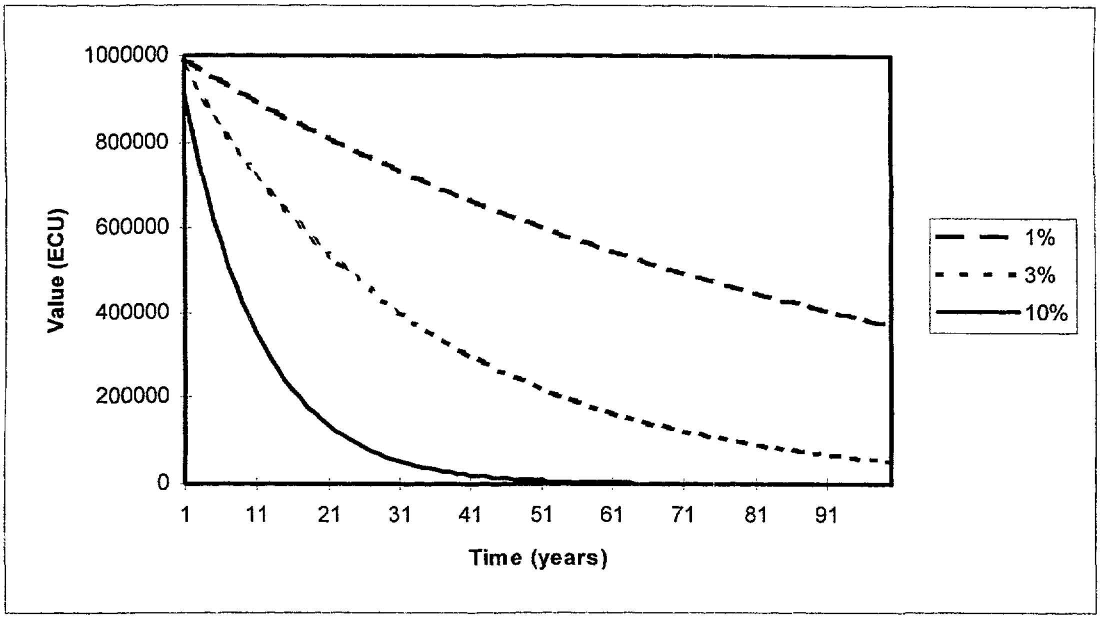

Figure V.1 Effect of discount rate on present value of damages worth 1 million ECU at the time (from 1 to 100 years in the future) when damage is incurred.

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

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

V.4. Results

Damages have been calculated for a range of different assumptions using both models. For the base case results shown in Table V.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 V.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 V.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 V.2.

Table V.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 V.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 V.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.

V.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 V.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 V.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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VI. VALUATION ISSUES

VI.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

VI.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.

VI.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.

VI.4. Transferability of Valuation Data

VI.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.

VI.4.2. Expert Opinion

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 carries this out. 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.

VI.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.

VI.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).

VI.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 form 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.

VI.6. Discounting

VI.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.]

VI.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. Krutilla (1967) has challenged this prescription at an intuitive level. For further discussions see Pearce and Markandya (1988) and Krautkraemer (1988).

VI.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.

VI.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.

VI.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.

VI.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. Economists in fact accept this argument, 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.

VI.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.

VI.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.

VI.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 could 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.

VI.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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VII. UNCERTAINTY AND SENSITIVITY ANALYSIS

VII.1. Introduction

In numerous places in this report it has been made clear that uncertainties in external costs analysis are typically large. 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 by statistical methods, allowing the calculation of formal confidence intervals around a mid 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.

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

VII.2. Analysis of Statistical Uncertainty

VII.2.1. Basis for the Analysis of 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 VII.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.

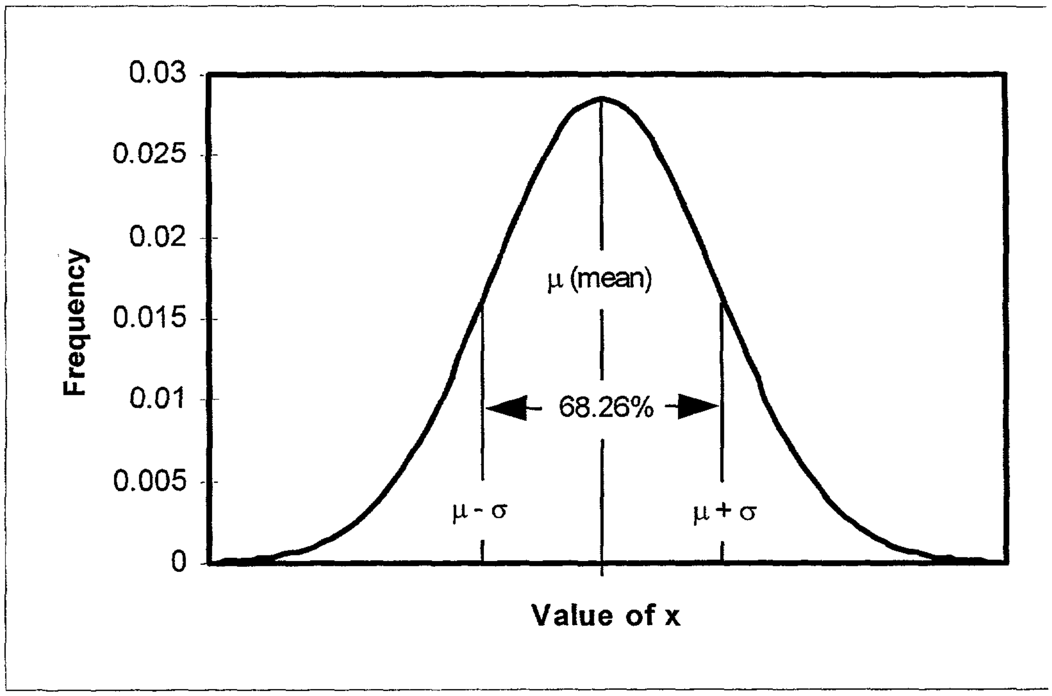

Figure VII.1 Illustration of the normal distribution.

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 (σ_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, σ_g approaches 1 and the lognormal distribution approaches the normal. In field sciences, both biological and social, larger uncertainties are common, so that $\sigma_g >> 1$.

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/σ_g to $\mu_g.\sigma_g$ and the 95% confidence limits by the range $\mu_g/\sigma_g{}^2$ to $\mu_g.\sigma_g{}^2$.

Acute mortality due to air-borne particulates is taken here as an illustrative example. There are three parts to the quantification of impacts;

- Estimation of emissions - for the macropollutants this is the best quantified stage of the analysis, with errors typically of the order of a few percent only.
- Dispersion - established models are available for describing the dispersion of pollutants around a point source, or from a number of different sources. The models are complex needing to integrate chemical processes and variations in meteorology over the extended distances over which they need to be applied. Overall, these models seem reasonably reliable, though it is difficult to validate output, and they are typically incapable of dealing with fine scale variation in pollution climate.
- Dose-response function - a number of epidemiological studies are available for assessing the acute effects of exposure to fine particles on mortality. Results are generally consistent.

From available information the geometric standard deviations for each step are estimated as;

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

the geometric standard deviation of the physical damage is $\sigma_g = 2.7$, from the formula:

$$\left[\log\left(\sigma_{g,tot}\right)\right]^2 = \left[\log\left(\sigma_{g,1}\right)\right]^2 + \left[\log\left(\sigma_{g,2}\right)\right]^2 + \left[\log\left(\sigma_{g,3}\right)\right]^2$$

for the combination of geometric standard deviations. If the median damage has been found to be $\mu_g = 2$ deaths/year, the one σ_g interval is 2/2.7 = 0.74 to 2*2.7 = 5.4 deaths/year, and the 95% confidence interval is $2/2.7^2 = 0.27$ to $2*2.7^2 = 14.58$ deaths/year. This result provides an indication of the likely range of outcomes based on statistical uncertainties, and an illustration of the shape of the probability distribution, skewed to the left, but with a long tail going out to high values.

In Table VII.1 the analysis is summarised and extended to include the errors arising through valuation. For this particular impact the valuation stage contains the most extensive uncertainties of all - a wide range of values have been suggested for the value of premature mortality linked to air pollution.

Table VII.1 Sample calculation of the geometric standard deviation for acute mortality due to air-borne particulates. Model, ethical and scenario uncertainties have been excluded from this analysis (see Section 3 of this Appendix)

Stage	**Geometric standard deviation σ_g**
Emission	1.1
Dispersion	2.5
Dose-response function	1.5
$\sigma_{g,tot}$ for impact assessment	2.7
Economic valuation	3.4
$\sigma_{g,tot}$ for cost	4.9
Effects not taken into account	>1.0
Grand Total σ_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.

VII.2.2. Confidence Bands

Estimates of σ_g (the geometric standard deviation) have been placed in three bands;

A = high confidence, corresponding to $\sigma_g = 2.5$ to 4;

B = medium confidence, corresponding to $\sigma_g = 4$ to 6;

C = low confidence, corresponding to $\sigma_g = 6$ to 12;

These bands are reported impact by impact elsewhere within this report. Given that σ_g has actually been quantified for a number of impacts (as in Table VII.1), it is reasonable to ask why the final result is given as a band. 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_γ^2. The overall ranges represented by the confidence bands are therefore larger than they might at first appear; band C covering four orders of magnitude.

VII.3. 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.

VII.4. Conclusions

The uncertainties involved in assessment of external costs can be very large - much larger than those experienced in many other disciplines. The reason for this is partly a function of the multiplicative nature of the analysis, and partly a function of the type of information used as input to the analysis.

Given these uncertainties it might be thought appropriate to question the validity of externalities analysis being used in relation to policy at the present time. However, if externalities analysis were abandoned, alternative means of informing policy makers would be required, and these would lack the following important attractions of the impact pathway approach;

- it provides a means of integrating information across disciplines
- results emerge at all stages of the impact pathway providing estimates for example of emission, population exposure, and extent of impacts, as well as monetary damages.
- the use of money for quantification of the final results provides an easily understood weighting system based on public preference.

The Appendix described the method developed by Ari Rabl and colleagues for the ExternE Project by which confidence bands have been derived for a number of the key impacts analysed in this study. Further details of the theory are provided in European Commission (1998). The method is based on the assumption that the probability distribution around some mid estimate is lognormal, reflecting the fact that most impacts are calculated by multiplying together a series of variables. The basic properties of the lognormal distribution were defined.

A problem arises because certain types of uncertainty are not amenable to statistical analysis - important issues surrounding discount rate and the future development of society. For these and similar parameters it is necessary to apply sensitivity analysis.

VII.5. References

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VIII. DEFINITION OF THE NATURAL GAS FUEL CYCLE, DATA AND RESULTS

This appendix gives the more detailed data concerning the natural gas fuel cycle. Only data, which have not been described in the main report, are described here. The natural gas fuel cycle is shown in Figure VIII.1.

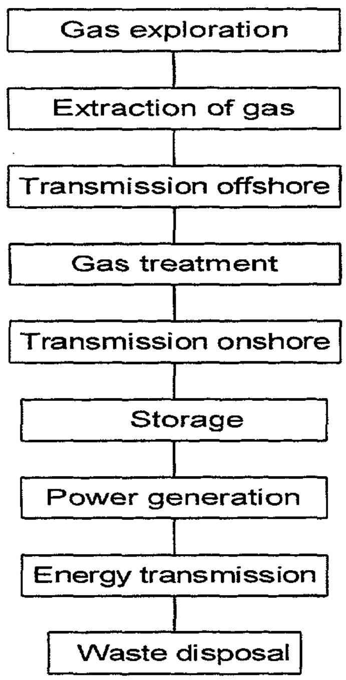

Figure VIII.1 The natural gas fuel chain for Denmark

The stages are described in the following table.

Table VIII.1 Definition of the natural gas fuel cycle

Stage	Parameter	Value
1. Gas exploration		
	Initiation	1984
	Methodology	Seismic exploration
2. Extraction of gas		
	Location	North Sea
	Distance to Jutland	300 km
	Annual production (1995)	$5x10^9$ m^3
	CO_2 emissions	73 g/kWh_{el}, 16.6 g/kWh_{heat}
3. Transmission offshore and gas treatment	Mode of transport	Pipeline
	Material	Carbon Steel
	Treatment	Pressurised, condensed and depressurised to 140 bar
		Dried by triethylenglycole
		Expansion to 80 bar
4. Transmission onshore and gas treatment	Mode of transport	Buried pipelines
	Material	Carbon Steel
	Treatment	Regulation to 16 or 40 bars
		Addition of tetrehydrotiophen
	CO_2 emissions	4.8 g/kWh_{el}, 1.1 g/kWh_{heat}
6. Storage		
	Site	Stenlille
	Type	Aquifer
	Storage volume	$300x10^{6\ m3}$
7. Power generation		
	Fuel	Natural gas
	Technology	Combined cycle CHP
	Location	Hillerød
	Installed power	77 MW_{el}, 75 MJ/s_{heat}
	Efficiency	44.4% el, 87.7% total
	Gas consumption	$78x10^6$ Nm^3 /yr
	Full load hours	3650 h/yr
	Lifetime	25 years
	Pollution control	Specific low NO_x turbine
	Air emissions	
	CO_2	460 g/kWh_{el}, 105 g/kWh_{heat}
	TSP	0.03 mg/kWh_{el}, 0.008 mg/kWh_{heat}
	SO_2	2.2 mg/kWh_{el}, 0.6 mg/kWh_{heat}
	NO_x	624 mg/kWh_{el}, 169 mg/kWh_{heat}
	CO	148 mg/kWh_{el}, 40 mg/kWh_{heat}
	Flue gas volume	500,200 Nm^3/h
	Flue gas temperature	374 K
	Height of stack	35 m
8. Energy transmission		
	electricity transmission	50-kV connection to consumers
	district heating transmission	Heat transmission system at a length of 31 km to 7 municipal district heating systems.
9. Waste disposal	Products	Domestic waste from the gas rig, construction vessels
		Oily wastes from supply and construction vessels
		Operational waste from construction activities
		Drilling fluids
		Material from pipelines/ decommissioning of plant

VIII.1. Hillerød CHP plant

Hillerød CHP plant is fully automatic. It may be started and stopped from the supervisor room, where regulation and supervision of the plant takes place. With the installed equipment it is possible to control the ordinary operation of the CHP plant from Kyndby plant. During remote control operation there is no manning at Hillerød CHP plant at night, but still there are two workers at the plant during the day.

Capacities and efficiencies of Hillerød CHP plant are shown in Table VIII.2.

Table VIII.2 Capacities and efficiencies for Hillerød CHP plant (SK Energi, 1994)

	Capacity	Efficiency
Electrical capacity	77.6 MW	
Heat capacity	75.6 MJ/s	
Heat storage	16,000 m^3 water	
Electrical efficiency		44.4 %
Total efficiency		87.7 %

The plant is a combined-cycle plant with both gas and steam turbines. The structure of the plant is shown schematically in Figure VIII.2.

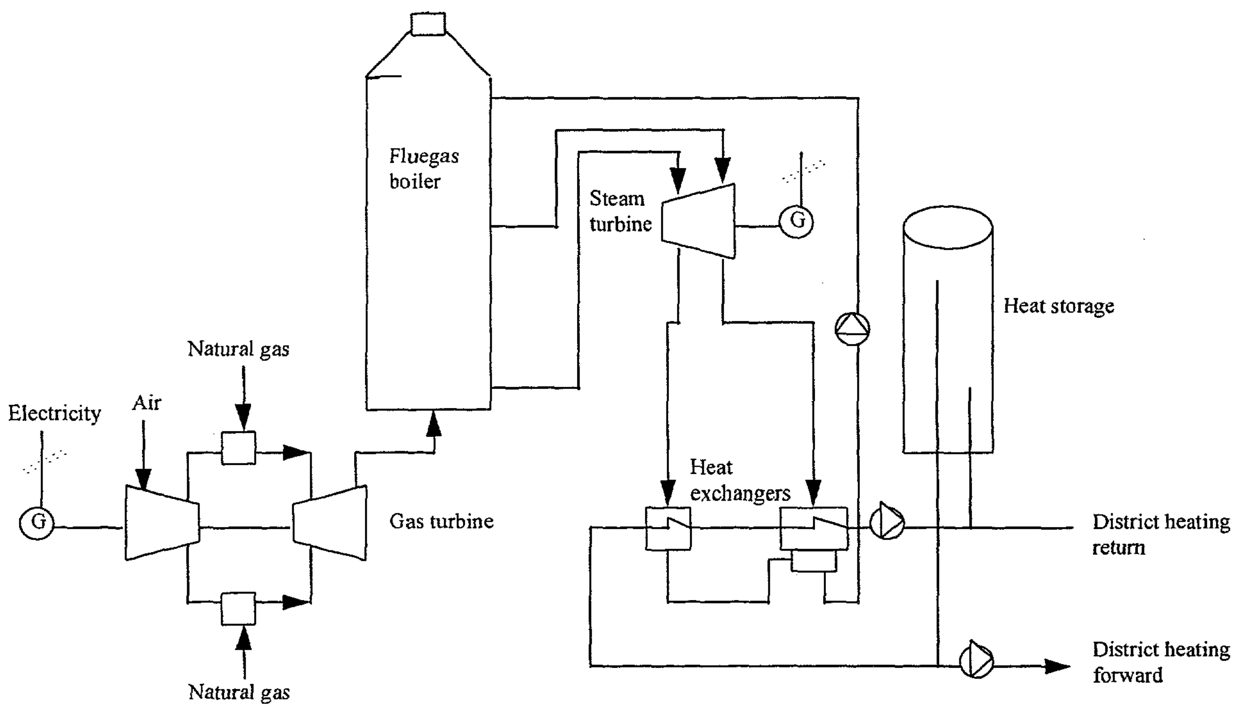

Figure VIII.2 Structure of the CHP plant in Hillerød

Air is drawn through large filters to the compressor, where it is compressed before natural gas is added and boiled. The exhaust gas flows to the gas turbine driving the generator, where the electricity is produced. The rest of the exhaust energy from the gas turbine is used in a steam boiler for producing steam, which drives a steam turbine connected to a second generator. In this way the steam increases the electricity production. The steam flows from the steam turbine to the heat exchangers, where the district heating water is heated

VIII.2. Burdens and impacts related to the natural gas fuel cycle

The burdens and impacts associated with the different steps of the natural gas fuel cycle are shown in Tables VIII.3-VIII.10. The final column in the tables shows the depth to which the impacts are analysed: "high" denotes impacts for which a quantitative analysis is performed, "medium" denotes impacts that are addressed qualitatively and "low" denotes impacts that are merely listed.

Table VIII.3 Impacts associated with gas exploration

Burden	Receptor	Impact	Priority
Occupational health:			
Accidents	Workers	Injuries, deaths	High
Atmospheric emissions:			
Emissions from exploration vessels	Numerous	Numerous	High
Emissions to marine environment:			
Debris	Fishing fleets	Loss of gear	Low
Oil, Chemicals, Drilling fluids	Marine life	Toxic effects	Low
Other burdens:			
Exclusion zone	Fishing fleets	Reduced catch	Low

Occupational health is related to a general risk at the working places and will be taken into consideration in most of the fuel cycle processes. For the exploration there is exposure of workers and ecosystem to radiation sources during seismic reflection surveys.

For gas exploration, atmospheric emissions are due to energy consumption from exploration vessels. Drilling is an energy-intensive activity, with energy supplied mostly by diesel generators. There will also be some venting and flaring of gas during the drilling operation. Emissions to air have been shown high priority.

Different drilling modes are used. The liquid base in the mud is composed of either sea water (water-based mud) or an emulsion of water in oil (oil-base mud). Oil-based modes are mainly used in the North Sea to avoid swelling of the clays, shales and mudstones through which the well passes. The drilling fluid is returned for reuse until its quality is too low. The waste mud is separated, but some oil remains in the cuttings, which means that some oil inevitably is discharged to sea. These impacts have a low priority.

Impacts associated with the construction and decommissioning of gas platforms, pipelines, gas treatment facilities, power stations, waste disposal sites and transmission lines, for the gas fuel cycle are shown in Table VIII.4. Occupational health is also here an important impact, which has been given high priority. Atmospheric emissions are due to the energy consumed during production of the materials and technologies. These emissions are given high priority.

Emissions to the marine environment are due to leakage during operation and leakage arising from decommissioning of the various technologies involved.

Other burdens involved include noise, physical presence of construction and deconstruction work, physical presence of vessels and vehicles, which have all been given low priority. The same applies for impacts due to the exclusion zone and disturbances of the ecosystem. Land use is due mainly to the establishment of the pipeline network where pipelines have been buried. The impact due to this construction work has been regarded as internalised even though the construction work disturbs traffic in the construction phase and may have a visual impact.

Table VIII.4 Impacts for the gas fuel cycle associated with the construction and decommissioning of gas platforms, pipelines, gas treatment facilities, power stations, waste disposal sites and transmission lines.

Burden	Receptor	Impact	Priority
Occupational health:			
Accidents	Workers and general public	Minor/minor injuries, deaths	High
Atmospheric emissions:			
Particulates	General public	Respiratory problems	Low
Secondary emissions	Numerous	Numerous	High
Emissions to marine environment:			
Debris	Fishing fleets	Loss of gear	Low
Oil, Chemicals	Marine life	Toxic effects	Low
Emissions from disused wells	Marine life	Toxic effects	Low
Other burdens:			
Noise	General public	Public nuisance	Low
Physical presence (onshore)	General public	Visual intrusion	Low
Physical presence	Shipping	Obstruction	Low
Exclusion zone	Fishing fleets	Reduced catch	Low
Disturbance	Ecosystems Agriculture,forestry	Reduced abundance	Low
Land use	Natural ecosystems	Loss of land	Internalised

Many of the impacts related to the operation of the platform are similar to those associated with exploration. The burdens and priority impacts are shown in Table VIII.5. In both cases high priority is given to atmospheric emissions, especially those of greenhouse gases. Furthermore, accidents due to the operation of the platform are given high priority. The same applies to the priority of impacts, which are shown in VIII.6a-b, due to operation of the pipelines. For gas treatment and storage other impacts than occupational health are also relevant. Emissions to air due to gas treatment and gas storage are given low priority, but any uncertainty due to the risk of the gas storage has been given high priority. The burdens and risks associated with this storage are seen in Table VIII.7.

Table VIII.5 Burdens and impacts associated with operation of the gas platform. Impacts related to global warming are addressed more fully in Tables VIII.8a to VIII.8d (impacts associated with the power generation stage). Such effects need not be considered separately for each stage of the fuel cycle because of the long atmospheric residence times of CO_2 and CH_4.

Burden	Receptor	Impact	Priority
Occupational health:			
Accidents	Workers	Minor/ major injuries, Death	High
Atmospheric emissions:			
Greenhouse gas emissions	General public	See Table VIII.8a	High
(CO_2, CH_4, and N_2O)	Ecosystems	See Table VIII.8b	High
Carbon monoxide	Human health	Respiration	Low
Other emissions	Numerous	Numerous	Negligible
Emissions to marine environment:			
Chemicals	Marine life	Toxic effects	Low
Produced waters	Marine life	Toxic effects	Low
Other burdens:			
Exclusion zone	Fishing fleets	Reduced catch	Low
Disturbance	Marine ecosystems	Abundance	Low

Table VIII.6a Burdens and impacts associated with operation of the offshore pipeline

Burden	Receptor	Impact	Priority
Occupational health:			
Accidents	Workers	Minor/major injuries	Medium
		Death	High
Emissions:			
Greenhouse gas emissions	Numerous	Numerous	High
Other burdens:			
Disturbance	Marine life	Abundance	Low
Exclusion zone	Fishing fleets	Reduced catch	Negligible

Table VIII.VIb Burdens and impacts associated with operation of the onshore pipeline

Burden	Receptor	Impact	Priority
Occupational health:			
Accidents	Workers and general public	Minor/major injuries	Medium
		Death	High
Emissions:			
Greenhouse gas emissions	Numerous	Numerous	High
Other combustion emissions	Numerous	Numerous	Negligible
Other burdens:			
Physical presence (compressor st.)	General public	Visual intrusion	Low
Disturbance	Ecosystems	Abundance	Low

Table VIII.7 Burdens and impacts associated with gas treatment and storage

Burden	Receptor	Impact	Priority
Occupational health:			
Accidents	Workers	Minor /major injuries	Medium
		Death	High
Emissions:			
Greenhouse gas emissions	Numerous	Numerous	Low
Other combustion emissions	Numerous	Numerous	Negligible
Other burdens:			
Physical presence	General public	Visual intrusion	Low
Physical presence (storage)	General public	Uncertainty	High
Disturbance	Coastal ecosystems	Abundance	Low
Noise	General public	Public nuisance	Low

Burdens and impacts associated with the power generation stage are divided in the following subcategories:

- Burdens and human-related impacts
- Burdens and impacts to terrestrial ecosystems
- Burdens and impacts to aquatic ecosystems
- Burdens and impacts to non-living systems
- Burdens and human related impacts

For these burdens it is especially the atmospheric emissions and occupational health which have been given high priority. These are the dominant burdens of the power-generation stage of the natural gas fuel cycle. Of other burdens the physical presence of the CHP plant has been given medium/high priority, as the plant situated in an area of natural beauty is visible for quite a distance.

Table VIII.8a Power-generation stage of the gas fuel cycle: Burdens and human-related impacts

Burden	Receptor	Impact	Priority
Occupational health:			
Accidents	Workers	Minor/major injuries	Medium
		Death	High
Atmospheric emissions:			
NO_x, CO, secondary particulates	General public	Respiratory symptoms	High
CO_2 and climate change	General public	Health effects	High
		Employment	High
	Low lying areas	Loss of homes/land	High
Other burdens:			
Noise	General public	Public nuisance	Low
Physical presence	General public	Visual intrusion	Med./High

Table VIII.8b Power-generation stage of the gas fuel cycle: Burdens and impacts to terrestrial ecosystems

Burden	Receptor	Impact	Priority
Atmospheric emissions:			
NO_x, precursors of O_3,	Forests	Direct effects on timber production	High
Acidity		Effects on tree appearance	Medium
		Interaction with pests	Medium
		Interaction with pathogens	Medium
		Interaction with climate	Medium
		Soil acidification	Medium
	Crops	Direct effects on yield	High
		Direct effects on quality	Medium
		Interaction with pests	High
		Interaction with pathogens	Medium
	Terrestrial	Interaction with climate	Low
	ecosystems	Direct loss of species	High
		Direct loss of habitat	High
	Agriculture	Sustainability	High
		Productivity	High
CO_2 (climate change)	Forestry	Sustainability (erosion)	Medium
		Productivity	High
	Natural	Sustainability	High
	ecosystems	Sustainability	High

Table VIII.8c Power-generation stage of the gas fuel cycle: Burdens and impacts to aquatic ecosystems

Burden	Receptor	Impact	Priority
Atmospheric emissions:			
NO_x, acidity	Rivers, lakes	Loss of fish and others	Low
		Sustainability	Low
CO_2 and climate change	Freshwater systems	Water availability	High
		Habitat loss	High
	Marine and estuarine systems	Water quality	High
		Habitat loss	High

Table VIII.8d Power-generation stage of the gas fuel cycle: Burdens and impacts to non-living systems

Burden	Receptor	Impact	Priority
Atmospheric emissions:			
NO_x, precursors of O_3, Acidity	Stones (in buildings)	Erosion/structural failure	Low
		Damage to cultural objects	No data
	Metals, Polymeric mat., fine art materials	Damage to cultural objects	No data
	Energy system	Changed demand	High
CO_2 (climate change)	Buildings	Subsidence	High
	Water supplies	Availability	High
	Buildings on low lying ground	Loss/damage through flooding	High

For burdens and impacts related to waste disposal, energy transmission and transportation of materials and personnel (VIII.IX-VIII.XI), only occupational health has been given high priority.

Table VIII.9 Burdens and impacts for waste disposal from the gas fuel cycle

Burden	Receptor	Impact	Priority
Occupational health:			
Accidents	Workers	Minor/major injuries	High
		Death	High
Atmospheric emissions:			
Dust, secondary emissions	Numerous	Numerous	Negligible
Emissions to water:			
Oils	General public	Water quality	Low
	Freshwater ecosystems	Toxic effects	Low
	Marine ecosystems	Toxic effects	Low
Other burdens:			
Land reclamation	Natural ecosystems	Creative conservation	Low

Table VIII.10 Transmission of electricity and heat. Issues related to land use were included in Table VIII.IV (impacts associated with construction and decommissioning).

Burden	Receptor	Impact	Priority
Occupational health:			
Accidents	Workers	Minor/major injuries,	High
		Death	High
Public health:			
Accidents	General public	Minor/major injuries	Low
		Death	Low
Electromagnetic fields	General public	Cancers	Not proven
Atmospheric emissions:			
Secondary emissions	Numerous	Numerous	Negligible
Other burdens:			
Physical presence	General public	Visual intrusion	Medium

Table VIII.11 Transport of materials and personnel

Burden	Receptor	Impact	Priority
Direct health effects:			
Accidents	Workers	Minor/major injuries	High
		Death	High
Vehicle emissions:			
Particulates, Ozone precursors,	General public	See Table VIII.8a	Medium
NO_x	Ecosystems	See Table VIII.8b	Low
	Forestry	See Table VIII.8b	Low
	Natural ecosystems	See Table VIII.8b	Low
CO_2	Numerous	See Table VIII.8a to d	Low
Emissions to marine traffic:			
Anti-fouling agents, Oil, Others	Marine life	Toxic effects	Low
Other burdens:			
Noise	General public	Public nuisance	Medium
Increased traffic	General public	Visual intrusion	Medium

VIII.3. Quantification of impacts and damages

The quantified impacts and damages related to the natural gas fuel cycle have been stated in chapter 3.4 of the main report. Here only the more detailed calculations, which have been done for some of the impacts are given.

VIII.3.1. Global warming effects of greenhouse gas emissions in relation to power generation

Greenhouse gases are emitted at many steps of the fuel cycle: the production of platforms, leakage from various sources, emission from transportation and production of energy.

The exploration process is an energy-intensive activity, as mentioned earlier. Approx. 6% of natural gas production is used for fuel consumption, while approx. 3% of the natural gas is flared. The emission due to fuel consumption from the North Sea in 1995 was 800 kt and flaring approximately 400 kt. The part related to the Hillerød CHP plant is 2.36%, corresponding to 19 kt CO_2 from fuel consumption and 9.4 kt from flaring. This figure is overestimated as a part of the gas was exported in 1995.

The high-energy consumption for extraction is due to new methods to extract the gas where large amounts of water are injected into the reservoirs at Dan, Gorm and Skjold (Danish Energy Agency, 1997). No figure has been available on energy consumption for compression. Using an energy consumption for compression of 0.5% of the gas production (CEC, 1995d), approximately 1.6 kt of CO_2 is emitted due to compression of natural gas to the Hillerød CHP plant.

It is estimated that approximately 650 t of materials were used in the manufacture of the CHP plant in Hillerød. All topside platforms in the Danish part of the North Sea comprises around 10,000 t of materials. Around 2.36% of this is related to the Hillerød CHP plant, giving 250 t materials for platforms. The weight of the jackets are, however, larger, up to 10 times, resulting in 2500 t related to Hillerød. Using a figure of 2 kg CO_2 per kg material (steel) produced in Denmark, a total emission due to production of platforms and power plant is 6.8 kt of CO_2. for the part related to Hillerød CHP plant. On an annual basis this leads to an emission of 0.27 kt/year. Production of pipelines and treatment plant on shore is not included. This amount is relatively negligible compared to the emission from the other fuel cycle processes.

In relation to the power generation at Hillerød CHP plant there are emissions of CO_2 from the combustion process at the plant. There are no emission data of CO_2 directly related to the plant, and therefore general data for decentralised burning of natural gas in Denmark are used. The emission factors used are shown in Table VIII.12.

Table VIII.12 Greenhouse gases related to power production based on natural gas (Fenhann, Kilde, 1994)

Greenhouse gases	Emissions factors
CO_2	56.9 kg/GJ
CH_4	0.025 kg/GJ
CO	0.020 kg/GJ
N_2O	0.001 kg/GJ

The emissions of N_2O, CH_4 and CO are converted to CO_2-equivalents by the following factors: 320, 21 and 1.4. The total greenhouse gas emissions in CO_2-equivalents for the power production stage are estimated to be 177 kt annually.

In relation to transmission and distribution of natural gas there are leakage of CH_4 resulting in greenhouse gas emissions to the atmosphere. According to a Nordic investigation (DGC/NGC, 1993) 0.02 % of the methane is emitted to the air during transmission and 1.34

% during distribution. Using these numbers for transmission and distribution of natural gas to the Hillerød CHP plant with a yearly utilisation of natural gas of $78*10^6$ Nm^3 (app. 71 Nm^3 CH_4) there is a yearly emission of $14*10^3$ Nm^3 methane due to transmission and $950*10^3$ Nm^3 methane due to distribution in the natural gas grid. This is approximately 11.2 t and 760 t CH_4, respectively (0.24 kt and 16 kt CO_2-equivalents). The emissions due to the distribution, however, is mainly due to the old town-gas grid and is therefore not included in the CO_2 balance for the Hillerød CHP plant.

The total emissions in CO_2-equivalents per kWh related to the Hillerød CHP plant is shown in Table VIII.13. 78% of the emissions have been allocated to electricity production and 22% to heat production.

Table VIII.13 Total emissions in CO_2 equivalents per kWh related to Hillerød CHP plant

	CO_2 emissions kt/year	g CO_2 /kWh $_{el}$	g CO_2 /kWh $_{heat}$
Exploration for gas	unknown		
Well drilling	unknown		
Offshore extraction	19	49	11
Flaring	9.4	24	5.6
Offshore pipeline leakage	negligible		
Transport of personnel to rig	negligible		
Liquid removal treatment	negligible		
Onshore compression	1.6	4.2	1.0
Onshore pipeline leakage	0.24	0.6	0.1
Production, construction and transportation of platforms and power plant	0.27	0.7	0.2
Transport of personnel to power plant	negligible		
Power generation	177	460	105
Total	208	539	123

VIII.3.2. Effects of atmospheric pollution in relation to power generation

Some technical and operational data of the plant are required to evaluate the external costs using the EcoSense system. These data are shown in Table VIII.XIV.

The operational data are from 1995.

Table VIII.14 Technical and operational data for Hillerød CHP plant, 1995

Gross electricity production:	248 GWh
Electricity send out:	240 GWh
Gross heat production:	258 GWh
Heat send out:	250 GWh
Full load hours:	3650 h
SO_2 emissions:	not measured
NO_x emissions:	105 mg/Nm^3
CO emissions:	not measured
Particulate emissions:	not measured
Stack height:	35 m
Stack diameter:	3.4 m
Flue gas volume flow:	500,200 Nm^3/h
Flue gas temperature:	374 K
Surface elevation at power plant site:	27 m

The flue gas volume flow is registered during a measurement of the NO_x emissions, in this way, volume flow and NO_x emissions correspond to each other.

SO_2, particulates and CO-emissions are not measured. For these the following emission factors are used:

SO_2 :	0.0003 kg/GJ	(Fenhann, Kilde, 1994)
Particulates:	0.00002 g/kWh	(CEETA, 1996)
CO:	0.020 kg/GJ	(Fenhann, Kilde, 1994)

Based on these factors and a natural gas consumption at the Hillerød CHP plant in 1995 at 2.27 PJ, the emissions to use as input for the EcoSense model are shown in Table VIII.15 and the total emissions in mg/kWh in Table VIII.16.

Table VIII.15 Estimated emissions in mg/Nm^3

SO_2 emissions:	0.37 mg/Nm^3
NO_x emissions:	105 mg/Nm^3
Particulates emissions:	0.005 mg/Nm^3
CO emissions:	24.9 mg/Nm^3

Table VIII.16 Total emissions of SO_2, NO_x , particulates and CO in mg/kWh

	emissions t/year	Emissions mg/kWh_{el}	emissions mg/kWh_{heat}
SO_2	0.681	2.2	0.6
NO_x	192	624	169
Particulates	0.0091	0.03	0.008
CO	45.4	148	40

The emission of SO_2 is so small that its use in ECOSENSE will give incorrect results, as it is close to the background level. Therefore, the emission of SO_2 is set to zero in ECOSENSE.

The total damages due to NO_x, CO and particulates are shown in the tables below. Only the mid estimates are shown in the tables, although for many of the damages there may be low and high estimates as well. The tables show the damages divided into those related to the electricity production and those related to the heat production, using the exergy approach.

Table VIII.17 Regional damages, mid estimate for crops

Receptor	Impact	Pollutant	$mECU/kWh_{el}$	$mECU/kWh_{heat}$
total	additional fertiliser needed (kg)	nitrogen deposition	0.00	0.00
total	additional lime needed (kg)	acid deposition	52e-4	14e-4
	TOTAL		52e-4	14e-4

The damage to crops is related only to acid deposition. The damage due to nitrogen deposition is negligible and on specific crops there are no damages as these are related to SO_2 .

Table VIII.18 Mid estimate for local, regional, and total damages for health in $mECU/kWh_{el}$ and $mECU/kWh_{heat}$

Receptor	Impact	Pollutant	Total damage (electricity)	Total damage (heat)
above 65 yrs	Congestive heart failure	tsp, nit, CO	0.03	0.01
adults	'Chronic' YOLL	tsp, nit	2.55	0.72
adults	Restricted activity days	tsp, nit.	0.08	0.02
adults	Chronic bronchitis	tsp, nit.	0.22	0.06
asthma, adults	Bronchodilator usage	tsp, nit.	0.01	24 e-4
asthma, adults	Cough	tsp, nit.	17e-4	5 e-4
asthma, adults	Lower respiratory symptoms	tsp, nit.	7e-4	2 e-4
asthma, child.	Bronchodilator usage	tsp, nit.	18e-4	5 e-4
asthma, child.	Cough	tsp, nit.	5e-4	2 e-4
asthma, child.	Lower respiratory symptoms	tsp, nit.	5e-4	1 e-4
children	Chronic cough	tsp, nit.	83e-4	22 e-4
children	Chronic bronchitis	tsp, nit.	64e-4	17 e-4
total	Respiratory hosp. admission	tsp, nit.	12 e-4	3 e-4
total	ERV for COPD	tsp, nit.	2e-4	0
total	ERV for asthma	tsp, nit.	1e-4	0
total	Hosp. visits child. Croup	tsp, nit.	5e-4	1 e-4
total	Cerebrovascular hosp. adm	tsp, nit.	29e-4	8 e-4
	TOTAL		2.91	0.81

The total damage given in Table VIII.18 is based on acute death calculated as years of life lost (YOLL).

Chronic YOLL as a result of nitrates and tsp is the dominant damage to human health accounting for about 88% of the total damage.

Table VIII.19 Regional damages in mECU/kWh, mid estimate for materials

Receptor	Impact	Pollutant	$mECU/kWh_{el}$	$mECU/kWh_{heat}$
galvanised st.	maintenance surface (m^2)	wet deposition	44e-4	12e-4
limestone	maintenance surface (m^2)	wet deposition	0	0
mortar	maintenance surface (m^2)	wet deposition	1e-4	0
natural stones	maintenance surface (m^2)	wet deposition	4e-4	1e-4
paint	maintenance surface (m^2)	wet deposition	0.03	87e-4
rendering	maintenance surface (m^2)	wet deposition	8e-4	2e-4
sandstone	maintenance surface (m^2)	wet deposition	0	0
zinc	maintenance surface (m^2)	wet deposition	2e-4	0
	TOTAL		0.04	0.01

Table VIII.19 shows the damage to materials as a result of wet deposition. These damages are only regional. The damage to painted surfaces is the dominant factor accounting for 85% of the damages on materials, while the damage to galvanised steel accounts for 11%.

VIII.3.3. Occupational and public accidents in relation to the whole fuel cycle

Occupational health

Offshore there have been only few fatal accidents during the last 15 years. In 1984 seven persons died due to a helicopter crash and in the end of the 1980s two more fatal accidents were reported. Only in spring 1997 has one more fatal accident occurred. This means that 10 fatal accidents have occurred during the last 15 years.

The number of injuries reported on average for the last 9 years is approx. 7 accidents per million working hours on fixed constructions and 12 per million working hours on mobile sources. This resulted in a total number of accidents of approximately 40 per year offshore (The Danish Energy Agency, 1996). Compared with figures for industry as a whole in Denmark (approx. 50 accidents per million working hours) these figures are relatively low. Data for Danish industry in general have been used in relation to the construction of the Hillerød power plant. The figure for constructing and establishing the natural gas grid (without the net for domestic natural gas supply) is considered to be small and has been neglected.

For the Hillerød power plant it is estimated that 40,000 person hours have been used in its construction (Elkraft), meaning that around 2 accidents had occurred during construction of the plant. The person hours for producing the materials and technologies are probably of the same magnitude. Using UK engineering sector figures fatal accidents are 0.162% of all accidents, major accidents are 12.9% and minor accidents are 87% of the total number of accidents.

VIII.3.4. Impacts specific to gas storage

The assessment of impacts related to gas storage is based upon information about the storage site Stenlille, located in mid-Zealand. Considerable efforts have been made to minimise the impact of the storage facility on the surrounding environment. The underground is constantly monitored to ensure that natural gas does not migrate from the storage area. As an additional safety measure, the groundwater supplying the Stenlille area with drinking water is controlled regularly.

The natural gas storage at Stenlille has a total volume of 300 mill. m^3. The storage facility consists of a central treatment plant, which is connected to three well sites by pipelines. The three well sites are located east of the village of Stenlille, approximately 1-2 km from the central plant. At each of the well sites a number of wells are drilled to reach the location in the underground, 1500 meters below ground, in which the natural gas is stored. In addition to the three well sites, a fourth has been reserved for possible drilling of new wells if future expansion of the storage facilities is required. Thus, there are a total of four well sites located around the central well.

The area extension of the underground storage is approximately 6 km^2. This area is shown in Figure VIII.3. Approx. 200 houses are located within it.

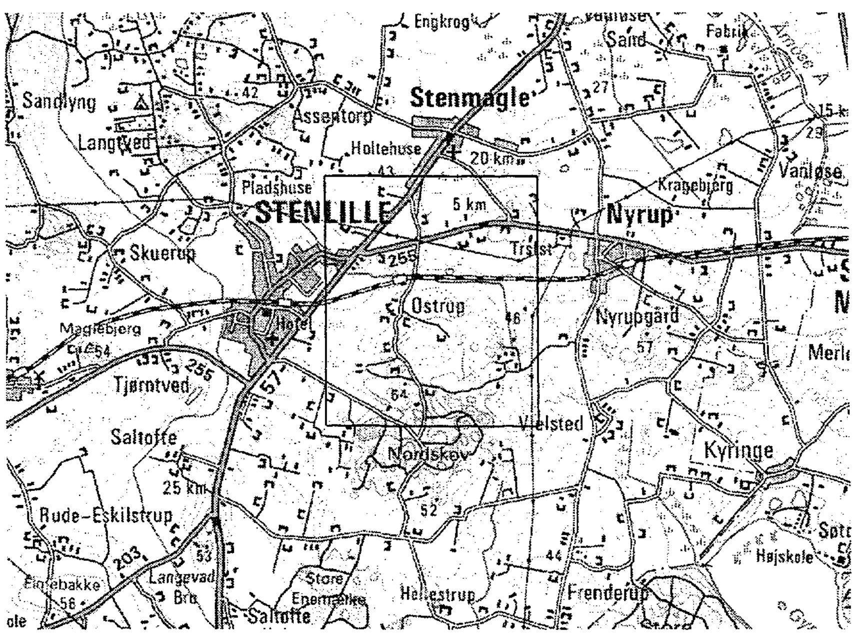

Figure VIII.3 Area extension of the underground storage in Stenlille

VIII.4. Total impacts and damages related to the natural gas fuel cycle

The quantified impacts and total damages which have been identified in relation Hillerød CHP plant are shown in Table VIII.20.

Table VIII.20 Total impact and damage in relation to the Hillerød CHP plant

Impact	Quantification	Monetisation, mECU/kWh $_{el}$	Monetisation, mECU/kWh$_{heat}$
Greenhouse gases	539 g CO_2/kWh$_{el}$ 123 g CO_2/kWh$_{heat}$	3.99	0.91
Atmospheric pollution	2.2 mg SO_2/kWh$_{el}$, 0.6 mg SO_2/kWh$_{heat}$	0	
	624 mg NO_x/kWh$_{el}$, 169 mg NO_x/kWh$_{heat}$	2.95	0.82
	148 mg CO/kWh$_{el}$, 40 mg CO/kWh$_{heat}$	0	0
	0.3 mg TSP/kWh$_{el}$, 0.4 0.008 mg TSP/kWh$_{heat}$	0	0
Ozone	624 mg NO_x/kWh$_{el}$, 169 mg NO_x/kWh$_{heat}$	0.57	0.15
Public accidents	negligible	0	0
Occupational accidents	24.13 minor accidents 3.58 major accidents 0.41 death	0.14	0.03
Marine environment	negligible	0	0
Natural gas storage	200 houses	1.0	0.28
Visual intrusion	25 houses	25e-4	7e-4
Land use changes	Negligible	0	0
TOTAL		8.65	2.19

VIII.5. References

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IX. DEFINITION OF THE BIOGAS FUEL CYCLE, DATA AND RESULTS

This appendix gives the more detailed data concerning the biogas fuel cycle. Only data, which have not been described in the main report, are described here.

IX.1. The biogas fuel cycle

In Figure IX.1 the fuel cycle is illustrated schematically.

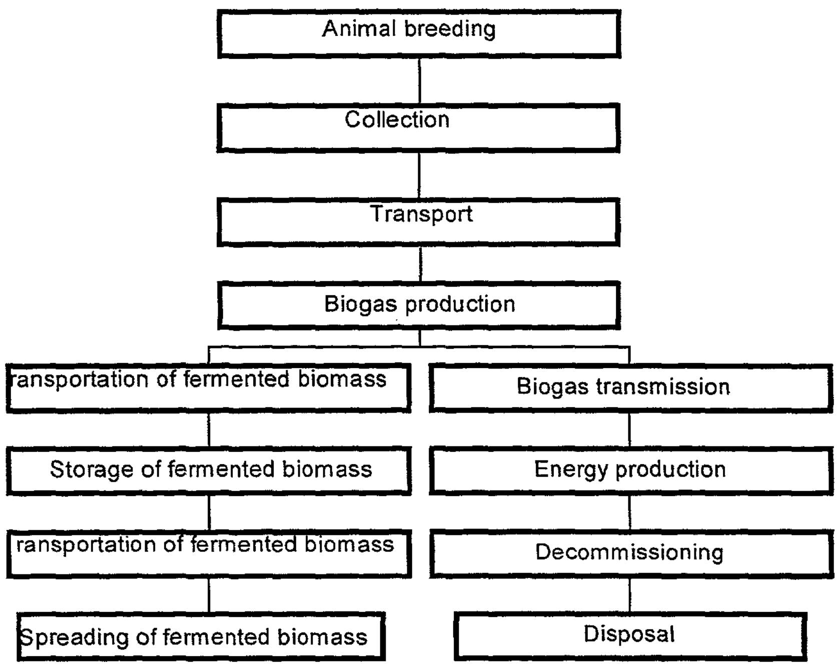

Figure IX.1 The biogas fuel cycle

The stages are described in the following table.

Table IX.1 Definition of the biogas fuel cycle

Stage	Parameter	Value
1. Animal breeding / food processing	Production of slurry for biogas plant Production of industrial organic waste for biogas plant	App. 330 tonnes/day App. 80 tonnes/day
2. Pumping and transportation	Location	Ribe/Esbjerg/Tønder
	Distance to biogas plant	Slurry app.11 km / indust. organic waste app 30km Trucks full load 18 t or 28 t per transport
	Mode of transportation	150,000 tonnes of biomass
	Annual transportation (1995)	153 m^3
	Annual fuel consumption	210,000 km
	Air emissions	
	CO_2	51 g/kWh_{el}, 14.3 g/kWh_{heat}
	TSP	36 mg/kWh_{el}, 10 mg/kWh_{heat}
	SO_2	64 mg/kWh_{el}, 18 mg/kWh_{heat}
	NO_x	670 mg/kWh_{el}, 190 mg/kWh_{heat}
3. Biogas production	Site	Ribe
	Material	Steel and concrete
	Daily production	12,000 m^3
	Site	Ribe Nørremark
	Type	Thermophilic digestion
	Year of construction	1990
4. Biogas transmission	Mode of transmission	Buried pipelines 2 km
	Material	Carbon Steel
	Treatment	Dried and regulated to 300 mbars
5. Power generation	Fuel	Biogas (1995: coal, 1997 natural gas as backup)
	Technology	Gas engine (1995: Caterpiller, 1997: Jenbacker)
	Location	Ribe Nørremark
	Installed power	1 MW_{el}, 5 MW_{heat} (1995, Including backup capacity)
	Efficiency	
	Gas consumption	34% el, 83% total
	Full load hours	$4.4 \cdot 10^6$ Nm^3/yr
	Lifetime	7000 h/yr
	Pollution control	15 years
	Air emissions	None
	CO_2	
	CH_4	0 g/ kWh_{el}, 0 g/ kWh_{heat}
	TSP	3.6 g/kWh_{el}, 1.0 g/kWh_{heat}
	SO_2	0.015 mg/kWh_{el}, 0.004 mg/kWh_{heat}
	NO_x	75 mg/kWh_{el}, 21 mg/kWh_{heat}
	CO	1350 mg/kWh_{el}, 350 mg/kWh_{heat}
	Flue gas volume	1350 mg/kWh_{el}, 350 mg/kWh_{heat}
	Flue gas temperature	6500 Nm^3/h
	Height of stack Annual electricity production	392 K 50 m
	Annual heat production	6,970 MWh (gross) 12,100 MWh (gross)

6. Storage of biomass at biogas plant	Site	Ribe
	Air emissions	
	CH_4	3,9 g/ kWh_{el}, 1.1 g/ kWh_{heat}
7. Pumping and transportation of biomass	Similar to pumping and transportation above	Included in "pumping and transportation" above
8. Energy transmission		
	Electricity transmission	50-kV connection to consumers
	District heating transmission	Heat transmission system at a length of 2-5 km to Ribe Nørremark.
9. Storage of biomass		
	Site	App. 11 km from the biogas plant in Ribe
	Number of sites	26 storage tanks
	Storage capacity	1-2,000 m^3 of biomass each
	Cleaning equipment	Layer of porous leca stones to avoid emission of NH3
10. Spreading of biomass	Site	Up to 15 km from the biogas plant on agricultural fields
11. Waste disposal		
	Products	Operational waste from construction activities Material from pipelines/ decommissioning of plant The fermented biomass is not regarded as waste

IX.1.1. Collection and transportation of biomass

Depending on the size of the biogas plant systems the slurry is transported to the biogas plant either by tractor or truck. In cases where the transportation distances are short the use of tractor is economical, but at larger distances, 5-10 km, larger trucks become advantageous. Two different kinds of transportation systems are available: unpressurised and pressurised tanks. The unpressurised tanks have the advantage of a larger capacity and lighter truck, whereas the disadvantages are wear and tear on pumps and the risk of damage to the pumps as a result of the impact of hard materials in the slurry. The pressure tanks are, on the other hand, more reliable. But they do have a lower capacity, due to a larger weight as well as to problems with production of "soap" at the pumping equipment (Danish Energy Agency, 1991a,b,c). In this way the two systems have their advantages and disadvantages, and at the beginning RBP uses the different systems on the various trucks for comparison. Today RBP uses unpressurised tanks.

IX.1.2. Production of biogas and gas treatment

Biogas is the product of an anaerobic biological process called methanogenesis. Biogas contains between 50% and 80% CH_4, and 15% to 45% CO_2. Furthermore, it contains about 5% water and traces of hydrogen sulphur and mercaptan. The advantage of producing CH_4 from the biomass compared to production of other products, i.e. alcohol, is that CH_4 is almost insoluble in the fermentation of mixed liquor and escapes spontaneously from the liquors without chemical treatment. There are, however, different options and obstacles for obtaining and optimising the biogas production. These regard control and removal of water, control of the H_2S production and different design options for controlling the biological processes.

The process of methanogenesis is the result of four consecutive steps: 1) solubilization-hydrolysis, 2) fermentation (or acidogenesis), 3) link processes, and 4) methanogenesis (Pauss, et al, 1987). The production of biogas is the result of a joint action of a rather large number of microbial species. The solubilization involves both a physical desegregation of the structure solid matter and a biochemical hydrolytic depolymerization. Typically the time needed to obtain the maximum solubilization may be up to two days and even up to tens of days depending upon the size of the solid matter. This influence the mean residence time of the solid matter in the bioreactor.

The energy content of biogas depends on the CO_2 content. The energy content (lower heating value) increases from 17.9 MJ/litre in a CH_4/CO_2 of a 50/50 gas to 28.7 MJ/litre in a 80/20 mixed gas of CH_4 and CO_2. The lower heating value of a 65/35 mixture is 23.3 MJ/litre, which is the value used for the energy content of the biogas in the present project.

Biogas typically contains up to 5% of water. This is important for the combustion efficiency of the biogas. If the combustion technology is able to recover the energy required to vaporise the water in the fuel the energy content of the fuel increases and drying of the biogas is unnecessary. The energy content in the fuel is then represented by the higher heating value. However, most small systems such as the gas engines for biogas combustion are unable to utilise the energy. Therefore, the lower heating value should be used as a measure of the energy content in the fuel - and the biogas should preferably be dried. Depending on temperature, biogas may contain up to 5% or 50 mg/l water vapour immediately after the outlet from the digester, which is near the saturation level (C.C: Ross, 1996). Water vapour in biogas can be removed by condensation, compression and/or cooling. The water content in the gas at RBP is almost negligible when it enters the CHP plant.

IX.1.3. Transmission of biogas and operation of pipelines

The biogas passes through a filter for removal of dust and afterwards the gas is transmitted directly to the pipeline through a absorption drier. In the event problems were to arise and the biogas should not be used at the CHP plant, the gas would then be transmitted to a booster compressor and stored in a high-pressure tank for some hours (Danish Energy Agency, 1991a,b,c).

Initially the gas was transmitted with a pressure of 250 kPa, but due to many technical problems the biogas is today transmitted with a pressure of only 30 kPa. The lower pressure reduces the electricity consumption, however.

IX.1.4. Storage of biomass

The normal way of handling the slurry in Denmark is to store it for up to 9 months in concrete storage tanks. The storage tanks are not necessarily equipped with lids. There is some focus on the emissions from storage tanks lacking lids, but the gas produced may still be very flammable and dangerous when handled unprofessionally. Some kind of storage technology or flaring technology might be developed in the future.

The storage capacity comprises the total biomass production in nine months from the farms and industries involved. In principle there are no differences between the stores at the farms and the intermediate stores, but the latter are handled more professionally taking the environmental impacts into consideration, by providing the storage with a layer of leca stones to reduce emissions of some of the pollutants.

IX.2. The biogas plant

Incoming biomass is pumped into three buffer tanks. The different biomass sources are kept separated until they are mixed in the reactor. One of the buffer tanks, the dosing tank, has been especially built to receive the organic waste and bleach soil, which is supplied to the reactor in doses to optimise biogas production. Experience has shown that this is the best way of controlling and optimising the biogas production. The tanks are equipped with lids with air filters to avoid an odour when loading. The predominant part of the sand in the manure will be separated in the buffer tank. The biogas plant is schematically illustrated in .

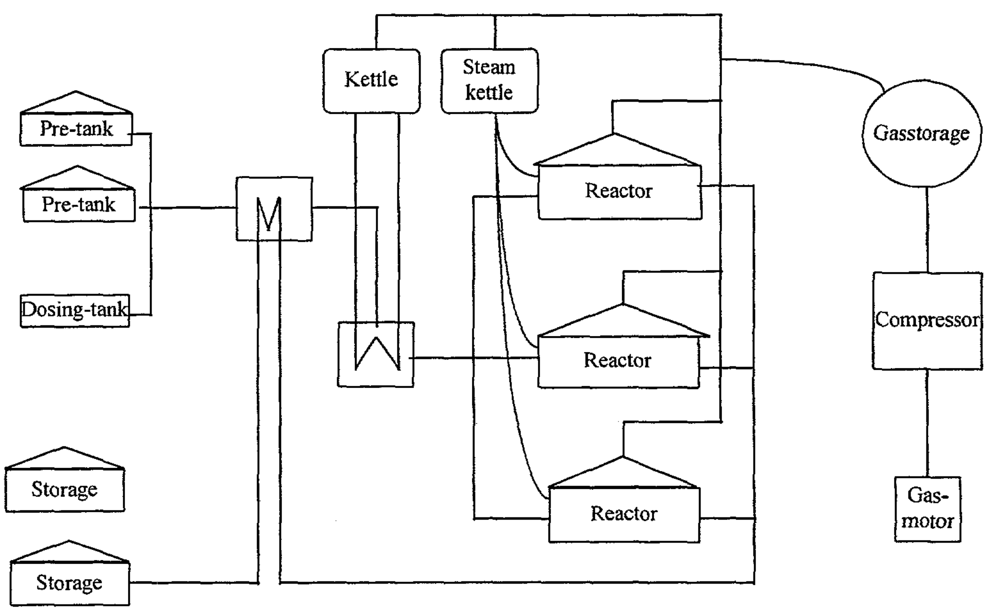

Figure IX.2 Ribe Biogas Plant

The two buffer tanks for slurry are equipped with rotors to obtain a homogeneous biomass before it is pumped into the reactors. The motors for the rotor systems are placed outside the tanks instead of lying within it. This has increased reliability considerably and it has a lower electricity consumption (Danish Energy Agency, 1991a,b,c). By mixing, any possible formation of flotation layers is avoided. Furthermore, the largest part of the sand in the slurry is separated at this stage. Before the biomass is pumped into the reactor it is comminuted into

1-2 mm particles in a macerator. This is done to obtain a smooth biomass to avoid clogging in the heat exchanger. The flow of biomass is measured in the pump system before it enters the reactor.

In Table IX.1 some of the technical data are summarised.

Table IX.2 Technical data of Ribe Biogas Plant

Biomass supply	410 t/day
Biogas production	12,000 m^3/day
Reactor volume	5,200 ^{m3}
Net biomass temperature	21.8 °C
Gas production used for process heating	12.6 %
Annual electricity consumption	697 MWh
Effectiveness of heat exchange	65 %
Storage capacity for biogas	1,000 ^{m3}

The fuel consumption for heating in a biogas plant is normally 10-15% of the biogas production. At RBP three are three different levels of heat supply/heat exchange in the biogas plant. The first heat exchange process is the exchange of heat from the outgoing fermented biomass with the incoming biomass. The second is heat exchange between the pre-heated incoming biomass and hot water produced in the steam boiler. The third heat supply also comes from the steam boiler, but in this case heat is injected into the reactor as steam to obtain the correct reactor temperature of 53°C. Optimisation in the heat exchange processes is an important way of optimising the biogas production, and further improvement is possible at the RBP.

IX.2.1. Power generation (Ribe-Nørremark combined heat and power plant)

The capacities and efficiencies of the CHP plant are shown in Table IX.II.

Table IX.3 Capacities and efficiencies for Ribe-Nørremark CHP plant (biogas part alone)

	Capacity	Efficiency
Biogas supply	3075 kJ/s (518 Nm3/h)	
Electrical capacity	993 kW	
Heat capacity	1814 kJ/s	
Electrical efficiency		32.3 %
Heat efficiency		59.0 %
Total efficiency		91.3 %

In order to evaluate the external costs using the EcoSense model, some technical and operational data of the biogas plant are required. These data are shown in Table IX.3.

Table IX.4 Technical and operational data for Ribe-Nørremark CHP plant, 1995

Electricity sent out:	6,973.2 MWh
Heat sent out:	12,140 MWh
Full load hours:	7000 h
SO_2 emissions:	25.7 mg/Nm^3
NO_x emissions:	290 mg/Nm^3
Particulate emissions:	0.0003 mg/Nm^3
Stack height:	50 m
Stack diameter:	0.5 m
Flue gas volume stream:	6,500 Nm^3
Flue gas temperature:	392 K

The emission of SO_2 is determined from the sulphur content in the biogas assuming that all H_2S is converted into SO_2. The emissions of NOx and particulates are determined as average figures for gas engines. The flue gas volume is estimated as an average figure for production of flue gas per GJ incoming fuel (natural gas).

IX.3. Overview of burdens related to the biogas fuel cycle

IX.3.1. Identification of impacts

IX.3.1.1 Collection and transportation of biomass

This part of the fuel cycle includes the transportation of biomass with trucks and pumping of biomass at the farms, pumping of the biomass at biogas plant and pumping of biomass either at the intermediate storage tanks or at the farms when delivering the fermented biomass.

There are large differences in transportation patterns among joint biogas plants, dependent on the use of intermediate storage tanks. In the case of RBP, intermediate storage facilities are incorporated in the transportation process. Environmental effects, however, do not vary remarkably. These effects are related to the transportation of slurry and are emissions to air due to the use of diesel fuel for the vehicles, noise from vehicles and risk for accidents related to the traffic of heavy trucks on the roads. These trucks also produce road damage due to wear and tear. All of these damages are almost no-existent when the slurry is not used for biogas production at joint biogas plants. Some transportation of slurry, however, does take place in any case, but this part of the fuel cycle is where the small farm-biogas plants have their advantages.

Pumping is carried out at the farm, at the biogas plant and at the intermediate storage while loading and unloading biomass. At the farm the biomass would be pumped in any case between the first buffer tank to the large storage tank. However, this pumping only occurs once. The pumping for use of the biomass at the biogas plant means that the biomass is pumped 4 or 6 times depending on whether the biomass is collected directly at the intermediate storage tanks for being spread on the fields (which is normally the case) or whether it is being transported to the store at the farm. In any case, the pumping is carried out by the trucks and emissions included for the emissions from transportation.

The emissions of NH_3, CH_4 and H_2S due to pumping and transportation are regarded to be negligible compared to the emissions related to the storage and spreading of biomass. The emissions of H_2S and NH_3, however, may create serious odour problems even though the emissions are very small. The emission at the farm is regarded to be negligible compared to the normal handling of slurry at the farm and the emissions at the biogas plant are negligible due to the loading and unloading in closed buildings and due to establishment of the biogas plant and storage tanks outside residential areas. The same applies to noise from pumping.

Impacts due to traffic noise are not considered as both the biogas plant and farms are situated outside urban areas and therefore the traffic will affect only few people. Impacts quantified due to transportation include only accidents and road damage (which will be studied in the next chapter) as well as exhaust emissions.

IX.3.1.2 Production of biogas and gas treatment (Ribe Biogas Plant)

Impacts related to biogas production include all emissions, which could possibly be emitted from biomass handling, namely emissions of NH_3, H_2S and CH_4 (emissions from combustion are treated later in relation to the combustion process). The emissions of NH_3 and H_2S are small and impacts are related only to offensive odours. As the plant is situated far from urban areas smells affect workers only at the plant. A visit to the plant shows that the impacts due to these odours are negligible. NH_3 and H_2S are hazardous only at very high concentrations, which is not the case at the biogas plant. Therefore, impacts due to these emissions are also neglected. With regard to the emission of CH_4, this will arise from storage tanks. This gaseous emission of will be further discussed and quantified in the next chapter.

The establishment of the biogas plant and the intermediate storage tanks has an influence on land use changes. The biogas plant is placed outside urban areas and affects very few people if at all. The 10-15 intermediate storage tanks are placed in the environment at strategically important places where more than one farm can collect the fermented biomass. If the biogas plant was not established, the slurry would be stored at the farms and the storage tanks built at the farms, being a part of the farm buildings they would therefore not be very harmful. But placed in the environment it might affect more people. The tanks are, however, placed far from urban areas and only in some cases would they be visible from bigger roads. They are mostly hidden behind trees and bushes. In this respect effects due to land use changes are neglected.

The mixing of slurry from different farms and also its mixing with industrial organic waste create some problems, mainly of a hygienic nature. Due to the possible risk of spreading diseases strict regulations have been enforced in Denmark. Specific demands on reactor time and temperature have been made. With these regulations the hygienic problems have been solved in Denmark today, and therefore these impacts are neglected.

IX.3.1.3 Transmission of biogas and operation of pipelines

The biogas is transmitted 2 km from the biogas plant to the CHP plant through 40 cm pipelines. The energy used for pumping, drying and compressing the biogas is included in the biogas plants own energy consumption (a total of 12.6% of biogas production). The transmission system comprises compressors, low pressure storage, absorption compressors, flaring stack, dust filter for filtering the biogas and eventually a high-pressure tank for emergency. Some of the water condensed in the pumping and absorption systems is transferred back to the biomass at the biogas plant. The rest is emitted to the waste water system.

At normal production there are no emissions due to leakage etc. The only emission to air is the flaring system. The gas emitted here is included in the overall energy consumption for the biogas plant. Filters are disposed of at disposal dumps. Impacts due to their disposal are neglected.

Energy consumption for maintenance of the transmission system is not included in the total figures of the energy consumption and is neglected.

IX.3.1.4 Power generation (Ribe-Nørremark CHP plant)

Damages from the power production are due mainly to the emissions from the combustion of biogas. Emissions produced are NO_x, SO_2, N_2O, CO, CH_4 and particulates. These emissions are a consequence of the combustion process itself. Emission of CO_2 is regarded to be neutral because the released CO_2 was absorbed from the atmosphere in the formation of the biogas.

IX.3.1.5 Production, construction and decommissioning of biogas plant, CHP plant, trucks, storage tanks and pipelines

Materials for the manufacture of the power plant comprise mainly concrete, steel and different non-steel metals. Most materials are produced at different industrial firms in Denmark and transported within the country. The use of materials and energy for the biogas plant are considerably higher than for the CHP plant due to the larger size of the former. Emissions from energy production for manufacture of the power plant will be calculated. The same regard emission from energy consumption for production of CHP plant, trucks, storage tanks and gas pipelines and the district heating system.

Other compounds are emitted to air directly in connection with the industrial production. These impacts are, however, very dependent on the various industries involved and the site of the industries, and are therefore not included. Including these impacts would need a very detailed study of the various impacts from the various industries involved. By neglecting them the local impacts where the materials are produced are underestimated.

The production of materials for the different parts of the fuel cycle also involves the production of wastewater and the consequent emissions to soil and water. These emissions are not included in the quantification and monetisation as is the case for emission to air from industrial production

Using average data for the industrial sector in Denmark includes impacts related to the working environment at the various industries.

Impacts studied due to construction of the biogas plant and the CHP plant are based only on emissions from energy consumption. Other impacts due to construction of the biogas and CHP plants are neglected.

Impacts related to decommissioning of the biogas plant, power plant and vehicles are also based only on the use of energy and emissions to air. Most materials are recycled, but some are disposed of, which means a risk of percolation of chemicals to the groundwater. However, in this case only emissions to air related to energy consumption are included and other impacts neglected.

IX.3.1.6 Disposal of wastes

The main "waste" at the biogas plant is the fermented biomass. This is, however, in this case not regarded as a waste, but as a valuable agricultural fertiliser. There will be emissions of CH_4, NH_3 and H_2S in relation to the storage and disposal of the biomass.

If the industrial organic waste were not delivered to the biogas plant it would be disposed of at a disposal dump and in a long time horizon the carbon in the waste would be transformed into CH_4 which, if uncollected, would be emitted to air as discussed further in the next sections.

No other waste from the biogas plant is produced except those from operation and maintenance, which has been neglected. This operation/maintenance waste arises from among other things the lack of cleaning at the combustion outlet for SO_2 or NO_x.

IX.3.1.7 Storage of biomass

If the biomass from the agriculture were not utilised for biogas production, the biomass would alternatively be stored, for example, in open storages at the farms. Some of the organic carbon would be transferred into CH_4 due to anaerobic digestion in the storage tank. The use of slurry for biogas production at the biogas plant reduces the emission of CH_4 from the agricultural sector. Therefore, it is decided to include this avoided emission in the overall balance of CH_4 emissions from the biogas fuel cycle. The normal way of handling the slurry in Denmark is to store it for up to 9 months in concrete tanks. It is then spread on the fields in spring. It is assumed that animal breeding is unaffected by this utilisation of slurry at the biogas plant.

The same argument could be applied to the use of industrial organic wastes for biogas production in the biogas plant. The organic waste would alternatively be disposed of at disposal dumps where the C would also be converted into CH_4. This conversion rate would be higher with the organic waste as it is considerable more easily digestible than the raw slurry.

By using the organic wastes at the biogas plant for co-fermentation with the slurry this emission of CH_4 at the disposal dump is avoided. This is also taken into account in the quantification of greenhouse gas emission of from the biogas fuel cycle.

When slurry is stored and spread NH_3 is emitted. The emissions of NH_3 depend on the constitution of the N (Organic N, NH_4^+) al well as on other factors such as pH.

Another factor of importance is that non-fermented slurry produces a floating layer in the storage tank, which reduces the emission of NH_3 by90%. This layer is not produced while storing fermented biomass. There are different possibilities for making artificial floating layers using straw or leca stones. Leca stones are used in the intermediate storage tanks at RBP. A layer of 10-20 cm of leca reduces the emissions of NH_3 by 96%.

IX.3.1.8 Spreading of biomass

Another factor of importance is the emission of NH_3 while spreading the biomass at the fields. This emission is produced in the same way as for storing the biomass. The emissions depend on the soil properties, soil constituents, which kind of machinery is used in the spreading and how fast the soil is treated afterwards. If the soil is harrowed immediately after spreading, the emission of NH_3 will be low, but just 4-12 hours waiting means a considerable emission of NH_3.

The emission of NH_3 is also in this case different for fermented biomass and non-fermented slurry. The pattern is the same as for emission from storages. One factor that reduces the emission of NH_3 from fermented biomass is that the fermented biomass is more fluent and will then wash down in the soil more easily. But still this factor is not as critical as, for instance, the soil treatment after spreading.

In the case where the raw slurry or the fermented biomass is spread on growing plants, the soil will not be treated afterwards and the emission will depend on the spreading method, the pH and how fast the biomass will percolate into the soil.

The emission of CH_4 from the soil after spreading is relatively low compared with the emissions from the storage tanks, as the slurry will dry fast and the aerobic process will very quickly take over, thus reducing the growth possibilities of the methane batteries.

Due to the digestion of raw slurry a large part of the organic nitrogen is converted into inorganic N. Much N evaporates as NH_3, as mentioned above, but a considerable fraction remains in the biomass as NH_4^+. A part of the N is converted into NO_3^- by identification. The amount of N-conversion into NO_3^-, which may percolate to groundwater, depends on the content of organic N.

IX.3.1.9 Transport of materials and personnel

The transport of biomass, which is the main transportation service in the biogas fuel cycle, had been described and discussed above. Other transportation demands take into consideration the transportation of workers from home to RBP and transportation of truck drivers from home to RBP if the trucks are parked at the biogas plant. The transportation of materials are also considered, such as delivery of spare parts, professional help from electricians and other technicians and delivery of diesel fuel for the trucks.

IX.3.2. Identification of impacts

IX.3.2.1 Collection and transportation of slurry, industrial organic wastes and digested biomass

The burdens and its connected impacts are shown in Table IX.4.

Table IX.5 Burdens and impacts associated with the collection and transportation of biomass

Burden	Receptor	Impact	Priority
Occupational Health:			
Noise	Workers	Hearing loss	Negligible
Physical Stress		Stress	Negligible
		Muscoloskeletal injury	Negligible
Atmospheric Emissions:			
Emissions from fuel	General public	Health effects	High
consumption:	Crops	Damage to crops	High
NO_x, SO_2, N_2O, CO_2 and	Forests	Damage to forests	High
particulates	Materials	Damage to materials	High
	Amenity (wetlands)	Impacts to ecosystem	Negligible
Public Health:			
Traffic	Pedestrians/cyclists/other	Minor, major injuries	High
	drivers/houses along	Deaths	High
	roads	Noise	High

IX.3.2.2 Production of biogas, transmission of biogas and storage and spreading of digested biomass (primary emissions)

The burdens, receptors, impacts and prioritised impacts are shown in Table IX.5.

Table IX.6 Burdens and impacts associated with production of biogas, transmission of biogas and storage and spreading of digested biomass (primary emissions)

Burden	Receptor	Impact	Priority
Occupational Health:			
Accidents	Workers (and general public)	Minor, major injuries	Negligible
		Deaths	High
Noise	Workers	Hearing loss	Negligible
		Stress	Negligible
Physical Stress	Workers	Muscoloskeletal injury	Negligible
Atmospheric Emissions:			
Emissions from biomass handling and leakage: NH_3, H_2S, CH_4	Forests	Acidification	High
	Human health	Psychological inconvenience	Negligible
	Workers		Negligible
	Damages due to the greenhouse effect	Greenhouse effect	High
	Neighbours	Unpleasant odours	Negligible
Emissions to soil:			
Percolation of pathogens	General public	Pollution of drinking water	High
Percolation of NO_3^-	General public	Pollution of drinking water	High

IX.3.2.3 Production of electricity and heat (primary emissions)

Table IX.6 shows burdens and impacts from energy production.

Effects on the wetlands west of Ribe are regarded to be small and therefore neglected. This area covers 3-6 km^2 and has important ecological and nature conservation values. The area is conserved mainly to protect the ecosystem and not for recreation. This means that even though people would never themselves be able to visit the area, there is nevertheless a willingness to give its conservation a high priority. The damages would be due to emissions to air from the CHP plant. However, in this case, the predominant wind is westerly, which means that only a small fraction of the emissions will affect the area. In this way the damages of the wetland due to emissions from the CHP plant are small and may be neglected. The issue is to be further discussed after evaluation of EcoSense results.

Table IX.7 Burdens and impact associated with the production of energy by biogas

Burden	Receptor	Impact	Priority
Occupational health:			
Accidents	Workers	Minor, major injuries	Negligible
		Death	High
Noise	Workers	Hearing loss	Negligible
		Psychological stress	Negligible
Physical stress	Workers	Muscoluskeletal injury	Negligible
Atmospheric emissions:			
Emissions from	Forests	Health effects	High
combustion: NO_x, SO_2,	Crops	Damage to crops	High
CO_2, CH_4, N_2O	Human health	Damage to forests	High
	Materials	Damage to materials	High
	Amenity	Impacts to ecosystem	Negligible

IX.3.2.4 Production, construction and decommissioning of the biogas plant, biomass storage tanks, transmission lines and CHP plant (secondary emissions)

Burdens related to this phase of the fuel cycle are shown in Table IX.VII.

Table IX.8 Burdens and impacts associated with production, construction and decommissioning of biogas plant, biomass storage tanks, transmission lines and power station for the biogas fuel cycle

Burden	Receptor	Impact	Priority
Occupational health:			
Accidents	Workers	Minor, major injuries	High
		Death	High
Noise	Workers	Hearing loss	Negligible
		Psychological stress	Negligible
Physical stress	Workers	Muscoluskeletal injury	Negligible
Atmospheric emissions:			
Secondary emissions: NO_x, SO_2, CO_2	Numerous	Numerous	High
Emissions to water:			
Release of waste water	General public	Water quality	Negligible
	Freshwater ecosystem	Toxic effects	Negligible
	Marine ecosystem	Toxic effects	Negligible
Other burdens:			
Noise	General public	Public nuisance	Negligible
Physical presence	General public	Visual intrusion	Negligible
Land use	Agricultural	Loss of land	Negligible

IX.3.3. Quantification of impacts and damages

IX.3.3.1 Impacts of atmospheric emissions

(1) Emissions from collection and transportation of slurry and industrial organic wastes

The total consumption includes transportation of both slurry and industrial organic wastes. The fuel consumption includes pumping of the biomass between stores and trucks. The fuel consumed at the farms in collecting the fermented biomass at the intermediate storage and bringing it to the fields is not included in the figures. The establishment of the biogas plant changes the nature of the farmers transportation pattern, but does not change the overall transportation demand for the farmers. A detailed study of the transportation demand shows that 40% of the farmers will get a reduced transportation demand and the rest will get no change in demand (Holm-Nielsen et al, 1993). However, the change in transportation demand for the farmers is regarded as negligible relative to the overall transportation demand from the biogas plant.

Table IX.VIII shows the damages on crops. The largest of these is due to additional lime needed due to acidification. The emission does have a benefit on fertilisation, however, which is around one-fourth of the value of adding lime. The total damages are only 0.0013 $mECU/kWh_{el}$ and 0.0004 $mECU/kWh_{heat}$.

Table IX.9 Regional damages on crops in mECU/kWh

Receptor	Impact	Pollutant	Damage $mECU/kWh_{el}$	Damage $mECU/kWh_{heat}$
Barley	Yield loss [dt]	SO_2	1e-4	0
Potato	Yield loss [dt]	SO_2	0	0
Sugar beet	Yield loss [dt]	SO_2	1e-4	0
Total	Add. fertiliser needed [kg]	acid	-3e-4	-1e-4
Total	Add. lime needed in kg	acid	12e-4	3e-4
Rye	Yield loss [dt]	SO_2	0	0
Oats	Yield loss [dt]	SO_2	0	0
Wheat	Yield loss [dt]	SO_2	2e-4	1e-4
Total			13e-4	3e-4

In Table IX.IX the damages on materials are shown. These figures include only regional damages. The largest damage is that to painted surfaces, this comprises around 80% of all damages to materials. The damages not only arise from SO_2, as shown in the table, but also from NO_x and the resulting acidification.

Table IX.10 Regional damages to materials in mECU/kWh

Receptor	Impact	Pollutant	Damage $mECU/kWh_{el}$	Damage $mECU/kWh_{heat}$
Galvanised st.	Maintenance surface (m^2)	SO_2	0.0062	0.0017
Limestone	Maintenance surface (m^2)	SO_2	0.0000	0.0000
Mortar	Maintenance surface (m^2)	SO_2	0.0002	0.0001
Natural stone	Maintenance surface (m^2)	SO_2	0.0000	0.0000
Paint	Maintenance surface (m^2)	SO_2	0.0291	0.0079
Rendering	Maintenance surface (m^2)	SO_2	0.0008	0.0002
Sand stone	Maintenance surface (m^2)	SO_2	0.0000	0.0000
Zinc	Maintenance surface (m^2)	SO_2	0.0001	0.0000
Total			0.0365	0.0099

The largest damages from the emission of SO_2 and NO_x are to human health. This is clearly shown in TableIX.X. The largest impact is on chronic mortality, here calculated as Years Of Life Lost (See Appendix.II.). The damage from chronic YOLL comprises 84% of the total damage to human health. In the table impacts on acute mortality are shown using the Value of Statistical Life of 3.1 mill. ECU, but the figure is not included in the total figures, as discussed in Appendix. The total damage is thus determined to be 2.95 $mECU/kWh_{el}$ and 0.80 $mECU/kWh_{heat}$ or 80 times higher than damage to materials and 2000 times higher than damage to crops.

Table IX.11 Total damages to human health in mECU/kWh

Receptor	Impact	Pollutant	Damage $mECU/kWh_{el}$	Damage $mECU/kWh_{heat}$
Above 65 years	Congestive heart failure	tsp, nitrate, sulfate	14 e-4	4 e-4
Adults	'Chronic' YOLL	tsp, nitrate, sulfate	2.53	0.69
Adults	Restricted activity days	tsp, nitrate, sulfate	0.08	0.02
Adults	Chronic bronchitis	tsp, nitrate, sulfate	0.21	0.06
Asthma adults	Bronchodilator usage	tsp, nitrate, sulfate	88 e-4	24 e-4
Asthma adults	Cough	tsp, nitrate, sulfate	17 e-4	5 e-4
Asthma adults	Lower resp. symptoms	tsp, nitrate, sulfate	7 e-4	2 e-4
Asthma child.	Bronchodilator usage	tsp, nitrate, sulfate	18 e-4	5 e-4
Asthma child.	Cough	tsp, nitrate, sulfate	6 e-4	2 e-4
Asthma child.	Lower resp. symptoms	tsp, nitrate, sulfate	5 e-4	1 e-4
Children	Chronic cough	tsp, nitrate, sulfate	82 e-4	22 e-4
Children	Chronic bronchitis	tsp, nitrate, sulfate	64 e-4	17 e-4
Total	Resp. hosp. admission	tsp, nitrate, sulfate, SO_2, NO_x	22 e-4	6 e-4
Total	ERV for asthma	tsp, nitrate, sulfate,	8 e-4	10 e-4
Total	Hosp. visits child. croup	tsp, nitrate, sulfate	2 e-4	1 e-4
Total	Cerebrov. hosp. adm	tsp, nitrate, sulfate	5 e-4	1 e-4
Total	'Acute' YOLL	SO_2, tsp, nitrate, sulfate, NO_x	29 e-4	8 e-4
Total			2.87	0.78

(2) Emissions from storage of fermented biomass

The system boundary for the fuel chain analysis includes estimates of avoided emissions of CH_4 from normal storage of slurry and organic waste. Some of these emissions are avoided when the CH_4 is produced in close reactor tanks at the biogas plant and the biogas is used for combustion.

The biomass, which is supplied to the biogas plant, is slurry from both pig farms and cattle-farms. At the Ribe Nørremark biogas plant around 20% of the slurry is from pigs, 60% from cattle farms and around 20% of the biomass is industrial organic waste. There is a difference in the composition of slurry from pig farms and that from cow farms. This difference results from the dissimilar digestive systems in the animals and their differing diets. However, in this study no difference in composition between the pig and cow slurry is assumed.

A Methane Conversion Factor (MCF) is defined to express the degree to which the carbon in the slurry is converted into CH_4 (UNEP, 1995). The emission of CH_4 from the slurry can vary from almost zero when the dung is dried "instantly" on the field to an MCF of 90% when slurry is stored in warm anaerobic lagoons. For systems where the slurry is stored in concrete tanks, which is the case for all Danish slurry production (Lauritsen, 1996), an MCF is estimated to be 10% in cool temperatures (<15°C), 35% in temperate climates (15-25°C) and 65% in warm temperatures (>25°C) (UNEP, 1995).

The slurry is first stored in small tanks at the farms up to 2 weeks before it is transported to the biogas plant when it is used for biogas production. When the slurry is not used for biogas production it is still stored some days in these small tanks before it is transmitted to larger ones. Therefore, the production and emission of CH_4 from the small tank is not taken into consideration as it is similar in the two cases.

In Denmark the greater part of the slurry is stored in large storage tanks during winter where the ambient temperature in some months is close to 0°C. Therefore, the temperature of the stored slurry will as an average be below 15°C. As the bacterial processes are anaerobic the slurry itself will not produce heat, and the temperature of the stored slurry, close to ambient temperature, will experience some temperature increase with the daily supply of fresh slurry.

(3) Emissions from production and transmission of biogas

It has been measured that 10% and maybe up to 20% of the CH_4 could be produced in the last storage tank where the biomass temperature falls due to a heat exchange with the incoming biomass. There is, however, still considerable bacteriological activities. If the CH_4 from the last storage tanks at the biogas plant is not collected it changes the greenhouse gas balance of the biogas plant considerably. UNEP (1995) estimates the MCF from a biogas plant to be 5-15%, which allows for large differences among biogas plants. If the CH_4 emission from the last stores is included, this figure seems to be relevant as general figures, but in the case of Ribe Biogas Plant, where the CH_4 from the last store is collected and the fermented slurry stays there for only 2 days, the figures seem to be too high. It is assumed in this case that only 2-3% of the original carbon is converted into CH_4 and emitted as leakage at the biogas plant (40,000 kg/year).

(4) Emissions from power generation

The results from the EcoSense runs are shown in Tables IX.11 to IX.13. Table IX.11 shows the damages to crops. The largest damage results from the addition of lime to obtain acidification. The damages are around 3 times higher than that from transportation. Here the benefit to fertilisation is also around one-fourth of the value of adding lime.

Table IX.12 Regional damages to crops in mECU/kWh

Receptor	Impact	Pollutant	Damages mECU/kWh$_{el}$	Damages mECU/kWh$_{heat}$
Barley	Yield loss [dt]	SO_2	4 e-4	2 e-4
Potato	Yield loss [dt]	SO_2	0	0
Sugar beet	Yield loss [dt]	SO_2	4 e-4	2 e-4
Total	Add. fertiliser needed [kg]	acid	-18 e-4	-4 e-4
Total	Add. lime needed [kg]	acid	68 e-4	18 e-4
Rye	Yield loss [dt]	SO_2	0	0
Oats	Yield loss [dt]	SO_2	0	0
Wheat	Yield loss [dt]	SO_2	1 e-4	2 e-4
Total			74 e-4	20 e-4

In Table IX.XII the damages to materials are shown valued as the cost of maintaining the surface. These damages include only regional ones. As in the case of damages from transportation, the largest are due to damage to paint comprising around 80% of all damages to materials. The damages are, however, not due to SO_2 alone as shown in the table, as the figures also include damages from acidification due to NO_x emission.

Table IX.13 Regional damages to materials in $mECU/kWh_{el}$

Receptor	Impact	Pollutant	Damage $mECU/kWh_{el}$	Damage $mECU/kWh_{heat}$
Galvanised st.	Maintenance surface (m^2)	SO_2	0.0366	0.0096
Limestone	Maintenance surface (m^2)	SO_2	0.0002	0.0000
Mortar	Maintenance surface (m^2)	SO_2	0.0012	0.0002
Natural stones	Maintenance surface (m^2)	SO_2	0.0002	0.0000
Paint	Maintenance surface (m^2)	SO_2	0.1740	0.0458
Rendering	Maintenance surface (m^2)	SO_2	0.0046	0.0012
Sandstone	Maintenance surface (m^2)	SO_2	0.0002	0.0000
Zinc	Maintenance surface (m^2)	SO_2	0.0008	0.0002
Total			0.2170	0.0570

The largest damages from SO_2 and NO_x emission are to human health, which is shown in Table IX.XIII. The largest impact is on chronic mortality, calculated as Year of Life Lost (see Appendix.). Chronic YOLL comprises 84% of the total damage to human health.

Table IX.14 Total damages to human health in mECU/kWh$_{el}$

Receptor	Impact	Pollutant	Damage mECU/kWh$_{el}$	Damage mECU/kWh$_{heat}$
Above 65 yrs	Congestive heart failure	tsp, nitrate, sulfate	78 e-4	20 e-4
Adults	'Chronic' YOLL	tsp, nitrate, sulfate	14.47	3.83
Adults	Restr. activity days	tsp, nitrate, sulfate	0.45	0.12
Adults	Chronic bronchitis	tsp, nitrate, sulfate	1.21	0.32
Asthma adults	Bronchodilator usage	tsp, nitrate, sulfate	0.05	0.01
Asthma adults	Cough	tsp, nitrate, sulfate	0.01	26 e-4
Asthma adults	Lower resp. symptoms	tsp, nitrate, sulfate	40 e-4	10 e-4
Asthma child.	Bronchodilator usage	tsp, nitrate, sulfate	0.01	26 e-4
Asthma child.	Cough	tsp, nitrate, sulfate	34 e-4	10 e-4
Asthma child.	Lower resp. symptoms	tsp, nitrate, sulfate	26 e-4	8 e-4
Children	Chronic cough	tsp, nitrate, sulfate	0.05	0.02
Children	Case of chron. bronchitis	tsp, nitrate, sulfate	0.04	0.01
Total	Resp. hosp. Admission	tsp, nit, sul, SO_2, NO_x	0.02	42 e-4
total	ERV for asthma	tsp, nit, sul, tsp, nit	12 e-4	2 e-4
total	Hosp. visits child. croup	tsp, nitrate, sulfate	24 e-4	8 e-4
total	Cerebrov. Hosp. adm	tsp, nitrate, sulfate	0.02	44 e-4
total	'Acute' YOLL	SO_2, tsp, nit, sul, NO_x	0.40	0.10
Total			16.74	4.44

(5) Production, construction and decommissioning of the biogas plant, biomass storage tanks, transmission lines and CHP plant (secondary emissions)

For renewable energy sources secondary emissions dominate the emissions from energy production. In the case of biogas production from slurry and organic waste, this domination is the case for some of the pollutants. The emission of CO_2 is predominant in the transportation stage, as the CO_2 produced in the combustion process is regarded to be zero. For SO_2 the production of the biogas plant is dominant, but for NO_x relatively small, 5-10% compared to the transportation and combustion processes. The emission of CO_2 is, however, around half of that from transportation. The energy consumption for producing the CHP plant is considered to be negligible compared with the production of materials to the biogas plant. The emissions of NO_x, SO_2 and CO_2 are shown in Table IX.14. Damages due to the production of technologies are not estimated. For NO_x and SO_2 together the emissions due to the production process is around 10% of the total emissions of NO_x and SO_2 for the transportation and combustion processes. This indicates the presence of a damage due to NO_x and SO_2 emissions from the production process of the biogas plant on the order of 10% of the damages from the transportation and combustion processes, on the order of 1 mECU/kWh$_{el}$ and 0.3 mECU/kWh$_{heat}$. These figures are, however, not included in the final tables.

Table IX.15 Secondary emissions due to production of materials and technologies for the biogas plant (Pedersen, P.B., 1991)

	CO_2	SO_2	NO_x
Total emission (kg/year)	224,000	1,300	923
Emission in g/kWh_{el}	21.8	0.128	0.090
Emission in g/kWh_{heat}	5.85	0.0345	0.024

IX.3.3.2 Road damage

For transporting the 320 t of manure per day and 23 t of biomass per transport, 6000 transports are carried out for transporting the slurry. The organic waste is transported over longer distances - with an average distance of 30 km from the biogas plant. Hence 1000 transports are carried out annually for carrying the organic waste. Estimated costs of maintaining the different road classes in West Jutland are shown in Table IX.XVII.

Table IX.16 Maintenance costs of Danish roads 1994. Includes all maintenance (grass cutting, cleaning, signals etc.) (Vejdirektoratet, 1997)

	Maintenance ECU/km
Highways	54,700
Large roads	18,300
Regional council roads	6,600
Local council roads	2,760

From the intermediate storage tanks tractors transport the manure, which also gives rise to road damage. As this is regarded to be neutral compared to the normal transport of manure this is not taken into account.

IX.3.3.3 Road accidents

For public accidents with regard to traffic it is assumed that two persons are employed at the biogas plant, each travelling an average transportation distance of 30 km per day, assuming that the worker stays either in Ribe or in Esbjerg. The total man-years are three for the operation and maintenance of the plant. Also three truck drivers are working full time. All of these will be driving to the biogas plant daily. This gives a total transportation of 36,000 km per year for the workers and truck drivers. The total transport distance of biomass is 210,000 km per year (1995).

Table IX.17 Total damage from the biogas fuel cycle

Impacts	Receptor	Damage mECU/kWh$_{el}$	Damage mECU/kWh$_{heat}$
Emission to air	Human health (CHP)	17.17	4.55
	Crops (CHP)	74 e-4	20 e-4
	Forests (CHP)	80 e-4	22 e-4
	Materials (CHP)	0.22	0.06
	Human heath (tr)	2.9453	0.80
	Crops (tr)	13 e-4	4 e-4
	Forests (tr)	15 e-4	4 e-4
	Materials (tr)	0.04	0.01
	N_2O	negligible	negligible
	NH_3	negligible	negligible
	H_2S	negligible	negligible
	CO_2	-0.244	-0.05
Emission to soil	Groundwater	negligible	negligible
Road damage	Road wear and tear	1.4	0.38
Accidents	Human health (tr.)	0.62	0.16
	Working environment	0.63	0.17
Total		23.12	6.07

CHP=combined heat and power

tr=transportation

IX.4. References

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X. DEFINITION OF THE WIND FUEL CYCLES, DATA AND RESULTS

This appendix gives the more specific data concerning the wind fuel cycle for the offshore wind farm Tunø Knob and the land based wind farm Fjaldene. Only data, which have not been described in the main report, are described here.

In order to include this chemical pollution the wind turbines are considered from a life cycle analysis (LCA) point of view. The life cycle has the following stages, as shown in Figure X.1.

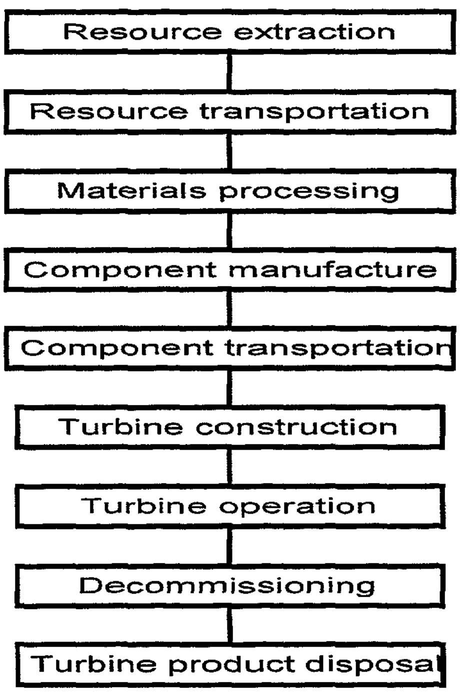

Figure X.1 Life cycle of the wind turbine fuel cycle

The wind farm analysed in the case study is an offshore wind farm consisting of 10 500 kW turbines with a total capacity of 5 MW.

For aggregation it has been necessary also to include a case study for an ordinary wind farm on land. The wind farm that have been chosen is a wind farm consisting of 18 500 kW turbines with a total capacity of 9 MW.

The details of the technologies assessed are shown in the following table.

Table X.1 Definition of the wind fuel cycles

Stage	Parameter	Value, off-shore	Value, on land
1. Turbine construction			
	Location	Roskilde	Roskilde
	Type	Vestas V39	Vestas V39
	Number of turbines	10	18
	Distance between turbines	200 m	188 m
	Distance between rows	400 m	580 m
	Characteristics		
	Rated power	500 kW	500 kW
	Rotor diameter	39 m	39 m
	Rotor speed	33 rpm	33 rpm
	Rated wind speed	16 m/s	16 m/s
	Tower height	40.5 m	40.5 m
	Weight	57 t	57 t
	Composition of turbines		
	Steel	527 t	949 t
	Aluminium	14 t	25 t
	Copper	3.5 t	6.3 t
	Sand	21 t	38 t
	Glass	11 t	20 t
	Plast	20 t	36 t
	Others	8 t	14.5 t
	Composition of fundaments		
	Reinforced iron	240 t	216 t
	Concrete	5650 t	5085 t
	Sea cables		
	Copper	25.8 t	
	Lead	33.6 t	
	Steel	39 t	
	PEX	5.4 t	
2. Turbine operation			
	Location	Tunø Knob	Fjaldene
	Power generation	12,500 MWh	19,800 MWh
	Lifetime	20 years	20 years
	Noise level	13.6 dB (A)	13.6 dB (A)

X.1. Tunø Knob offshore wind farm

Tunø Knob wind farm is located at a northern geographical latitude of 55.57 degrees and an eastern longitude of 10.21 degrees. The wind farm consists of 10 wind turbines. The turbines are three-bladed Vestas V39 offshore pitch-regulated machines, each with a capacity of 500-kW at a nominal wind speed of 16 m/s. The tower height is 40.5 m and the rotor diameter is 39 m. The detailed technical data of Tunø Knob wind farm is shown in Table X.2

Table X.2 Technical data for Tunø Knob wind farm (Madsen, 1996)

Generator capacity	5 MW
Number of turbines	10
Type	Vestas V39 500-kW offshore
Power control	pitch regulation
Hub height	40.5 m + 2.5 m foundation
Rotor diameter	39 m
Rotor speed	33 rpm
Operation range	4-25 m/s
Weight	57 t
Monitoring	radio communication
Arrangement	two straight lines north-south
Distance between turbines	200 m
Distance between rows	400 m
Distance to shore	6000 m
Water depth	3.1 - 4.7 m
Foundation type	box caisson
Foundation weight	1000 t
Expected yearly net electricity production	12,500 MWh

The wind farm is connected to the grid via a 6 km sea cable to Saksild Beach. In the village of Saksild the wind farm is connected to the high-voltage grid in East Jutland via a 60/10 kV transformer.

The wind farm is placed with the turbine fundaments at sea level. The height of the turbines is 40.5 m. The wind farm has 10 turbines and occupies an area of approximately 32 hectares. The turbines are placed in two parallel rows from north to south with a distance of 400 m between the rows. The distance between the wind turbines in the row is 200 m (Midtkraft, 1994). The nearest village is Tunø village 4.3 km to the east. The nearest town is Odder with about 9,500 inhabitants at a distance of 13 km west of the wind farm.

The offshore wind turbines look like ordinary wind turbines, but the specific conditions at sea have made some changes necessary:

- Normally wind turbines on land have transformer and main switch placed separately beside the turbine. In the Tunø wind turbines both parts are placed on the turbine foundation in the bottom of the tower. This provides better protection against salt water and high sea. At the same time the temperature inside the tower and the rotor hat is increased due to the waste heat from the transformer. A higher temperature inside the tower protects the inner parts against corrosion.

- Crane equipment is installed inside each of the turbines, so that all components including gearbox and generator may be replaced without the use of a floating crane.
- The doors to the towers are raised in order to avoid a covering of ice in the winter.
- The towers are surface protected against corrosion with an extra thick cover of topcoat.
- The colour of the turbines (the marine-grey colour of the Navy) is darker than normal. In this way they are less visible.

X.2. Fjaldene wind farm

Fjaldene wind farm is located at a northern geographical latitude of 56.9 degrees and an eastern longitude of 8.34 degrees. The wind farm consists of 18 wind turbines placed in two rows with 9 turbines in each row. The distance between the rows is 580 m, while the distance between the wind turbines in the row is 188 m. Each turbine has a capacity of 500 kW. The height of the turbines is 41.5 m.

The detailed technical data of Fjaldene wind farm is shown in Table X.3.

Table X.3 Technical data for Fjaldene wind farm

Generator capacity	9 MW
Number of turbines	18
Type	Vestas V39 500 kW
Power control	pitch regulation
Hub height	41.5 m
Rotor diameter	39 m
Arrangement	two straight lines
Distance between turbines	188 m
Distance between rows	580 m
Height above sea level	83 m
Expected yearly net electricity production	19,800 MWh

X.3. Overview of burdens related to the wind fuel cycle

The most relevant environmental burdens due to the full life cycle of the wind turbines and the associated distribution system are summarised in Table X.4 to X.V. The final column in the tables shows the depth to which the impacts are analysed: "high" denotes impacts for which a quantitative analysis is performed, "medium" denotes impacts that are addressed qualitatively and "low" denotes impacts that are only listed.

Table X.4 Impacts associated with the operation of offshore wind turbines

Burden	Receptor	Impact	Priority
Occupational Health			
Accidents	Workers	Minor/major injuries, deaths	Low
Public Health			
Accidents	Public, boats	Minor/major injuries, deaths	Medium
Amenity Impacts			
Noise	Residents, others	Noise amenity	High
Visual intrusion	Residents, visitors at neighbouring coasts	Visual amenity	High
		Flicker annoyance	Medium
Scattering of radio waves	Radio users	Radio interference	Medium
Ecological Impacts			
Sea use	Natural ecosystems	Change in current conditions	Low
	Shells and fish	Death, injury or disturbance	High
Turbine motion	Birds	Death, injury or disturbance	High

X.3.1.1 Occupational Health

Burdens included in Occupational Health are accidents, noise and physical stress. The impacts, all of which apply to the workers at the wind farm, have low priority, as the wind farm is visited only by technicians for inspection of the wind turbines twice a year, resulting in a very low probability for accidents. The wind turbines are operated from the operational central office at Midtkraft, and run fully automatically.

X.3.1.2 Public Health

Accidents are the only burdens included in Public Health. Accidents in the shape of wind blades flying off may cause minor or major injuries or even death to people at a distance from the turbines. As the turbines at Tunø Knob are located at sea the only potential danger would be to sea voyages. The probability that a person in a boat would be struck by a wind blade is almost negligible, and therefore accidents to public health are given low priority.

Accidents may also happen in the road transportation of workers at the wind farm. The wind farms are operated in a remote-control mode from Midtkraft and Vestkraft, and the road transportation relates to the movement of workers from home to the site in the operation of the wind farms at Midtkraft and Vestkraft.

X.3.1.3 Amenity Impacts

Burdens included in Amenity Impacts are noise, visual intrusion and scattering of radio waves. Noise from the wind farm is a burden to the residents and other people in the area close to the wind farm. As Tunø Knob wind farm is located at sea 3 km from land the noise effect is negligible. Still, as noise is the most discussed burden in relation to wind energy this burden is given high priority. Also in the case of Fjaldene wind farm the assessment of noise is quite important.

Visual intrusion is a burden for residents, visitors, travellers and others near the wind farm. The region is a popular one for summer residents and visual intrusion is therefore a burden that has caused a lot of discussion. The visual burden is therefore given a high priority.

Scattering of electromagnetic waves may cause interference for radio and TV users in the vicinity of the wind farm. Residents in the area may not be affected, but scattering of radio waves may be a problem to sailors in the area. Scattering of radio waves is therefore given medium priority.

X.3.1.4 Ecological impacts

Burdens included in Ecological Impacts are sea use and turbine motion. The utilisation of areas at sea for the siting of Tunø Knob wind farm may affect the natural ecosystem and fishes and shells in the area. Current conditions near the wind farm may be changed and the life of shells and fishes in the area may also be changed. Threatened nature types on the seabed may disappear as a consequence of the establishment of the offshore wind farm.

Investigations have been made of the current conditions in the area around Tunø Knob based on earlier hydraulic investigations. The result of these investigations is that the existence of the offshore wind farm will not affect the current conditions. Therefore this impact is given a low priority.

The motion of the turbines may cause death, injury or disturbance of birds close to the wind farm. Tunø Knob is located in an area between two larger Ramsar areas with resting eiders at the islet and large passages of birds over the islet. Therefore the effect of the blade rotation is given a high priority.

Table X.5 Impacts associated with the non-operational stages of the life cycle of offshore wind farms

Burden	Receptor	Impact	Stages assessed	Priority
Occupational Health	General public	Injuries, deaths	Construction [2]	High
Accidents	Workers	Injuries, deaths	Manufacture	High
Accidents		Injuries, deaths	Others	Medium
Public Health				
Emissions [1]	General public	Respiration	Materials processing and manufacture	High
Greenhouse gases [1]	General public	Food shortage etc.		High
Amenity Impacts				
Noise	Residents, others	Noise amenity	All	Low
Visual intrusion	Residents, others	Visual amenity	All	Low
Ecological Impacts				
Sea use	Natural ecosystems	Change in steam conditions	Construction	High
	Shells and fish	Death, injury or disturbance		High
Emissions [1]	Agriculture, forestry	Production	Materials processing and manufacture	High
	Terrestrial ecosys fisheries	Various		Medium
Greenhouse gases [1]	Agriculture	Production		High
	Ecosystems	Various		Medium
Other Impacts				
Particulates [1]	Materials	Cleaning	All	Low
Greenhouse gases [1]	Water supply, Materials damage	Various	Materials processing etc.	Medium

[1] *The quantitative assessments of these impacts are based on work on the external costs of the coal and lignite fuel cycles from the ExternE project, to which the reader should refer for more details. Even within this reference a detail assessment of the impacts of global warming is not possible.*

[2] *Accidents due to transport of construction workers between home and work.*

Table X.6 Impacts associated with the electricity distribution systems for offshore wind turbines

Burden	Receptor	Impact	Priority
Occupational Health			
Accidents	Workers [1]	Minor/major injuries, deaths	High
Public Health			
Accidents	General public	Minor/major injuries, deaths	Low
Electromagnetic radiation	General public	Human health	Medium
Amenity Impacts			
Noise	Residents, others	Noise amenity	Low
Visual intrusion	Residents, others	Visual amenity	High
Earth movement	Natural ecosystems	Land loss	Low
Sea cables	Fishes	Injury or disturbance	Low
Ecological Impacts			
Electrified cables	Birds	Death, injury or disturbance	Medium

[1] *Accidents in construction and maintenance of the electricity distribution equipment on site are not separated in the analysis from the accidents due to comparable activities on the turbines themselves.*

The impacts identified concerning the operation of offshore wind turbines are the same as the impacts identified for an ordinary wind farm on land.

Also, the impacts associated with the non-operational stages of the life cycle are the same for offshore wind farms as for ordinary wind farms except for the ecological impacts. While one of the ecological impacts for an ordinary wind farm is earth movement, the corresponding burden for an offshore wind farm is sea use causing changes in current conditions and death, injury or disturbance of fish and shells.

Impacts associated with the electricity distribution systems are the same for offshore wind turbines as for ordinary wind turbines except for amenity impacts, where sea cables are included as a burden for the offshore wind farm.

X.3.2. Quantification of impacts and damages

This chapter specifies some of the data used for quantification of impacts and damages in chapter 5.4 in the main report.

X.3.2.1 Noise

Noise level and its effect are calculated by a logarithmic formula, which includes the distance from the wind turbine (CEC, 1995f). The formula is adjusted for the variation between night and day sensitivity, irregular operation, noise sensitivity of people and background noise. The formula for the annual value of noise, AVN, is as follows:

$$AVN = \sum_{allpositions} (L_{year,obs} - L_{dn,back}) * N_{houses} * A(P) * NDSI$$

Where

$L_{year,\ obs}$ =	Average of noise whilst the turbines are in operation over a period of a year
$L_{dn,\ back}$ =	Expected noise without the turbines
N_{houses} =	Number of houses at that location
A(P) =	Annuitised average house price
NDSI =	Noise depreciation sensitivity index

In the case of the Tunø Knob wind farm noise calculations have been made as well for Tunø, located 3 km from the wind farm as for Saksild Beach located 6 kilometres from the wind farm. At Tunø the noise level is found to be about 14.5 dB(A), while the noise at Saksild Beach is 1.5 dB(A), indicating that only houses at Tunø will be disturbed by the turbines. At a distance of 3500 meters the noise level from the wind farm is about 11 dB(A) (Ministry of Environment, 1991). Taking into account that the noise level from rattling leaves closely is 10 dB(A) only houses at Tunø at a distance of 3000 to 3500 meters from the wind farm are considered. In this area about 22 summer residents and 1 farm are located.

The following values are used to calculate the annual value of noise:

$L_{year,\ obs}$ =	13.6 dB(A)
$L_{dn,\ back}$ =	10 dB(A)
N_{houses} =	22 summer residents, 1 farm
P =	House prices
	41,000 ECU (300,000 DKr) for summer residents (Denmark Statistics, 1995)
	172,000 ECU (1,250,000 DKr) for farms (Denmark Statistics, 1995)
A =	5 % (the market discount rate announced by the Ministry of Finance)
NDSI =	0.5 % (European Commission, 1997)

By using these numbers the annual value of noise is calculated to be 967 ECU for Tunø Knob wind farm or 0.004 mECU/kWh.

Assuming that the noise level for Fjaldene wind farm is the same as that for Tunø Knob the noise level from the wind farm is about 11 dB(A) at a distance of 3500 meters (Ministry of Environment, 1991) and only houses in this distance from the wind farm are therefore considered. In this area about 90 family houses and 30 farms are located.

The following values are used to calculate the annual value of noise for Fjaldene wind farm:

$L_{year,\ obs}$ =	13.6 dB(A)
$L_{dn,\ back}$ =	10 dB(A)
N_{houses} =	90 houses, 30 farms
P =	74,000 ECU (550,000 DKr) for family houses (Denmark Statistics, 1995)

	172,000 ECU (1,250,000 DKr) for farms (Denmark Statistics, 1995)
A =	5 % (the market discount rate announced by the Ministry of Finance)
NDSI =	0.5 % (European Commission, 1997)

By using these numbers the annual value of noise is calculated to be 10,640 ECU for Fjaldene wind farm assuming that all the houses are influenced by a noise of 13.6 dB, which is not the case. It is assumed that half of the houses are influenced by 13.6 dB, while the rest are influenced by 11.5 dB. Using these numbers the annual value of noise is calculated to be 7,535 ECU for Fjaldene wind farm or 0.019 mECU/kWh.

X.3.2.2 Visual amenity

A Danish study has been carried out assessing the visual effect and noise from wind turbines (Jordal-Jørgensen, J., 1995). In the study three surveys have been carried out:

- A survey of the number of houses directly affected by the wind turbines
- An interview survey giving the willingness to pay getting rid of the wind turbines
- A house price survey evaluating the influence on the prices of houses near wind turbines

The interview survey has shown that 13% of the people living in the vicinity of wind turbines are bothered by the turbines and the willingness to pay to get rid of the wind turbines, for those who were able to express it that way, were on average 132 ECU per household per year. In relation to the electricity production the cost is largest for single turbines (0.2 mECU/kWh), while for clusters it is 0.1 mECU/kWh and 0.03 mECU/kWh for wind farms.

A survey of house prices has shown a systematic tendency for houses, which are affected by wind turbines on the purchase date, to be cheaper than other houses. The effect on the houses is shown in Table X.7.

Table X.7 The effect of wind turbines on the house prices

Type of plant	Effect on house price (ECU per plant)	Effect per wind turbine	Number of observations
Single turbines	2,135	2,135	6
Clusters	16,500	4,715	7
Wind farms	12,655	1,055	3

The conversion of the difference in house prices is made based on the costs associated with the finance of the increased price of those houses unaffected by wind turbines. Included in these costs are interest and repayment of loan minus tax benefit. These costs are subtracted from the extra price of the house, when the house is sold again.

The results from the survey of house prices are very uncertain, as they are based on a limited number of observations. The results for wind farms are especially uncertain as only three observations were made.

The effect on house prices of houses in the vicinity of a wind farm is used as monetisation value for the Fjaldene wind farm. The effect on the house price is related to noise as well as visibility of the wind farm. Noise has already been monetised and the effect on the house price will therefore as an estimate be halved to take only the visibility into consideration.

X.3.2.3 Impacts of atmospheric emissions

The total amount of material for Tunø Knob and Fjaldene wind farms is shown in Table X.8.

The amount of material per wind turbine used for the fundaments of the wind farm on land is assumed to be half the amount used for a wind farm offshore.

Table X.8 Materials used for Tunø Knob and Fjaldene wind farm

Materials	Tunø Knob	Fjaldene
Turbines		
Steel	527,000 kg	948,600 kg
Aluminium	14,000 kg	25,200 kg
Copper	3,500 kg	6,300 kg
Sand	21,000 kg	37,800 kg
Glass	11,000 kg	19,800 kg
Plast (polyester and epoxy)	20,000 kg	36,000 kg
Oil products	1,000 kg	1,800 kg
Others	7,000 kg	12,600 kg
Fundaments		
Reinforced iron	240,000 kg	216,000 kg
Concrete	5,650,000 kg	5,085,000 kg
Sea cables		
Copper	25,800 kg	-
Lead	33,600 kg	-
Steel	39,000 kg	-
PEX	5,400 kg	-

The energy use and atmospheric emissions in relation to the above given amount of materials are quantified based on Danish studies (Fenhann, J., and Kilde, N.A., 1994) (Schleisner, L. et al, 1995). The total energy use for the specific materials is shown in Table X.9.

The energy use is related to production, transportation and manufacture of one kg material, and has not been calculated for sand, oil products and concrete. As one approach the energy use for glass has been used for these materials.

Table X.9 Total energy use for specific materials

	Coke (MJ/kg)	Coal (MJ/kg)	Oil (MJ/kg)	Natural gas (MJ/kg)	Total (MJ/kg)
Steel	1.6	14.1-20.7	4.9-8.2	0.1	20.7-30.6
Aluminium	0	23.1-31.5	8-11.4	1.4-2.9	32.5-45.8
Copper	3	45.1	13.6	16.5	78.2
Sand	0	1	0.8	7.5	9.3
Glass	0	1	0.8	7.5	9.3
Plast (polyester and epoxy)	0	30.8	9.8	5.1	45.7
Oil products	0	1	0.8	7.5	9.3
Reinforced iron	7.4	10.6	18.2	0.1	36.3
Concrete	0	1	0.8	7.5	9.3
Lead	0	20.3	9	6.3	35.6
PEX	0	30.8	9.8	5.1	45.7

The emission factors used are as follows:

Table X.10 Emission factors (Fenhann, J., and Kilde, N.A., 1994), kg/GJ

	SO_2	NO_x	CO_2	N_2O	CH_4	CO
Electricity production (coal)	0.714	0.400	95.0	0.003	0.0015	0.010
Coke combustion	0.680	0.200	102.0	0.003	0.0015	0.097
Gas oil combustion	0.094	0.100	74.0	0.002	0.0015	0.012
Natural gas combustion	0.0003	0.150	56.9	0.001	0.0040	0.013

The emissions from the production of Tunø Knob wind farm have been divided into those due to electricity and heat related to the production of the materials and those related to transportation of the materials (see Table X.11).

Table X.11 Emissions for production of Tunø Knob divided into electricity, heat and transportation emissions

	SO_2 (kg)	NO_x (kg)	CO_2 (kg)	N_2O (kg)	CH_4 (kg)	CO (kg)
Electricity emissions	15680	8283	2118253	66	33	462
Heat emissions	663	7232	2992287	57	185	650
Transport emissions	2735	8076	432430	18	11	2607
Total	19078	23591	5542970	141	229	3719

The emissions estimated apply to plants without desulphurisation plants or de NO_x burners, and the emissions are therefore a high estimate. The heat emissions are especially related to the production of concrete to the fundaments. The SO_2 and NO_x emissions due to electricity production will be reduced 50%, as many plants in Denmark are provided with desulphurisation plants reducing the SO_2 emissions by about 80%, and de NO_x burners reducing the NO_x emissions by about 70%.

The emissions related to electricity production are based on an average coal-fired plant located in Denmark. Data for the coal-fired plant Fynsværket are used together with Danish meteorological data for production of wind turbines and other materials. Data used as input in EcoSense are the following:

Table X.12 Data used for EcoSense

Capacity:	400.0 [MW]
Electricity sent out:	366.0 [MW]
Full load hours per year:	7300 [h]
SO_2 Emissions:	304.0 [mg/Nm3]
NO_x Emissions:	580.0 [mg/Nm3]
TSP Emissions:	24.0 [mg/Nm3]
(CO emissions:	11.19 [mg/Nm3])
Stack height:	235.0 [m]
Stack diameter:	5.0 [m]
Flue gas volume stream:	1168000.0 [Nm3/h]
Flue gas temperature:	351.0 [K]
Surface elevation at site:	15.0 [m]
Anemometer height:	235.0 [m]
Geographical latitude:	55.41 degree
Geographical longitude:	12.08 degree

Fynsværket is in this way located in St.Valby (55.41, 12.08) close to Roskilde, where the meteorological data have been measured. The results from the EcoSense runs have been scaled down regarding the emissions, corresponding to the emissions related to Tunø Knob offshore wind farm and Fjaldene onshore wind farm. The CO emissions are based on the CO emission factor for electricity based on coal.

The total damage for Tunø Knob is shown in the following tables.

Table X.13 Regional damages (Tunø Knob) in mECU/kWh, mid estimate for crops

Receptor	Impact	Pollutant	Mid damage
Barley	yield loss [dt]	SO_2	0.00
Potato	yield loss [dt]	SO_2	-1 e-4
Sugar beet	yield loss [dt]	SO_2	0.00
Total	additional fertiliser needed (kg)	Nitrogen deposition	0.00
Total	additional lime needed (kg)	Acid deposition	2 e-4
Rye	yield loss [dt]	SO_2	0.00
Oats	yield loss [dt]	SO_2	0.00
Wheat	yield loss [dt]	SO_2	1 e-4
	TOTAL		2 e-4

The damage to crops is a result of nitrogen and acid deposition, acid deposition being the dominant factor. In Table X.13 SO_2 fertilisation has been taken into account. If the SO_2 fertilisation is neglected the total damage for crops would be 0.0016 mECU/kWh.

The damage to crops is related only to acid deposition. The damage due to nitrogen deposition is negligible and on specific crops there are no damages as these are related to SO_2.

Table X.14 Regional damages (Tunø Knob) for health in mECU/kWh

Receptor	Impact	Pollutant	Mid damage
Above 65 yrs	Congestive heart failure	tsp, nit., sul., CO	18 e-4
Adults	Chronic YOLL	tsp, nit., sul.	0.37
Adults	Restricted activity days	tsp, nit., sul.	0.01
Adults	Chronic bronchitis	tsp, nit., sul.	0.03
Asthma, adults	Bronchodilator usage	tsp, nit., sul.	12 e-4
Asthma, adults	Cough	tsp, nit., sul.	3 e-4
Asthma, adults	Lower resp. symptoms	tsp, nit., sul.	1 e-4
Asthma, child.	Bronchodilator usage	tsp, nit., sul.	3 e-4
Asthma, child.	Cough	tsp, nit., sul.	1 e-4
Asthma, child.	Lower resp. symptoms	tsp, nit., sul.	0.00
Children	Chronic cough	tsp, nit., sul.	12 e-4
Children	Chronic bronchitis	tsp, nit., sul.	9 e-4
Total	Resp. hosp. admission	tsp, nit.,sul	2 e-4
Total	Resp. hosp. admission	SO_2	2 e-4
Total	ERV for COPD	tsp, nit., sul.	0.00
Total	ERV for asthma	tsp, nit., sul.	0.00
Total	Hosp. visits child. croup	tsp, nit., sul.	0.00
Total	Cerebrov. Hosp. adm	tsp, nit., sul.	4 e-4
Total	'Acute' YOLL	SO_2	47 e-4
	TOTAL		0.42

The total damage given in Table X.14 is based on acute death calculated as years of life lost (YOLL). Using value of statistical life (VSL) would give the following results, Table X.15.

Table X.15 Mortality based on VSL

Impact	Pollutant	Mid damage
Chronic mortality	tsp, nit, sul	1.37
Acute mortality	SO_2	0.13
TOTAL		1.50

Chronic mortality calculated as years of life lost is the most dominant damage to health. The damage is a result of nitrates, sulfates and particles; by far the largest part is from nitrates (about 2/3).

Table X.16 Regional damages (Tunø Knob) for materials in mECU/kWh

Receptor	Impact	Pollutant	Low damage	Mid damage	High damage
Galvanised st.	maintenance surface (m^2)	SO_2 , wet dep.	13 e-4	17 e-4	24 e-4
Limestone	maintenance surface (m^2)	SO_2 , wet dep.	0.00	0.00	0.00
Mortar	maintenance surface (m^2)	SO_2 , wet dep.	1 e-4	1 e-4	1 e-4
Natural stone	maintenance surface (m^2)	SO_2 , wet dep.	0.00	1 e-4	1 e-4
Paint	maintenance surface (m^2)	SO_2 , wet dep.	68 e-4	86 e-4	0.01
Rendering	maintenance surface (m^2)	SO_2 , wet dep.	2 e-4	3 e-4	4 e-4
Sandstone	maintenance surface (m^2)	SO_2 , wet dep.	0.00	0.00	1 e-4
Zinc	maintenance surface (m^2)	SO_2 , wet dep.	0.00	0.00	0.00
	TOTAL		0.01	0.01	0.02

Table X.16 shows the damage to materials. The highest damage is that to painted surfaces, accounting for about 76% of the total damage to materials, whereas the damage to galvanised steel accounts for about 14%.

X.3.2.4 Accidents

Public accidents

Tunø Knob is an offshore wind farm, which is operated from Midtkraft outside Århus. Therefore assumptions must be made about numbers of people at Midtkraft working in relation to the wind farm, the length of journey etc. The offshore wind farm is inspected only twice yearly. The number of road traffic accidents is related to the amount of kilometres from home to work, and are standard assumptions. Accidents at sea are neglected.

The operation from Midtkraft outside Århus occupies work of about 5 minutes a day, corresponding to a total of 3 man days pr. year. People working at Midtkraft are supposed to live in Århus. The transportation distance from Århus to Midtkraft is 5 km each way, making a total amount of 30 km pr. year.

The offshore wind farm has service inspections twice a year. A service inspection occupies 5 men in 5 days, a total amount of 50 man days pr. year. Beside this there are unforeseen accidents, assumed to occupy 5 men in 10 days. Altogether a total amount of 100 man days pr. year are used at the wind farm. The transportation distance to the wind farm from Århus is 20 km each way, corresponding to a total amount of 4000 km pr year for service inspection and unforeseen accidents. This number of kilometres may be lower than this as two or three of the workers may be travelling together.

Using the above-mentioned number the total distance related to the operation of Tunø Knob wind farm is 4030 km pr. year.

Fjaldene wind farm is operated from Vestkraft in Esbjerg. The operation from Vestkraft occupies work of about 5 minutes a day, corresponding to a total of 3 man days pr. year. People working at Vestkraft are supposed to live in Esbjerg. The transportation distance from Esbjerg to Vestkraft is 5 km each way, making a total amount of 30 km pr. year.

The wind farm has service inspections twice yearly. A service inspection occupies 4 men in 5 days, a total amount of 40 man days pr. year. Beside this there are unforeseen accidents, assumed to occupy 4 men in 5 days. Altogether a total amount of 60 man days pr. year are used at the wind farm. The transportation distance to the wind farm from Esbjerg is 75 km each way, corresponding to a total of 9000 km pr year for service inspection and unforeseen accidents.

Using the above-mentioned number the total number of km related to operation of the wind farm is 9030 pr. year.

Also, road transportation in relation to the whole life cycle are included in the calculation of public accidents. For the constructional phase for Tunø Knob the work consumed about 135 man months. The total amount of km related to construction of the wind farm is 36,000 km. For Fjaldene the construction work (based on information concerning Tunø Knob) is assumed to consume 36 man months with a total distance related to construction of the wind farm of 24,000 km.

The following accident data are estimated from statistical information over the years 1990-1994 (Automobil-importørernes sammenslutning, 1995) (Denmark Statistics, 1995):

- Accidents pr. million km of transportation: 0.15
- Killed pr. million. km of transportation: 0.009

Using the above-mentioned accidents the number of accidents and death in relation to the wind farm can be estimated as in Table X.17. The accidents have been divided into minor and major accidents (Denmark Statistics, 1996).

Table X.17 Public accidents

	Tunø Knob (offshore)		Fjaldene (on land)	
	Daily operation	Construction	Daily operation	Construction
Minor accidents	0.0051	0.0023	0.0114	0.0015
Major accidents	0.0070	0.0031	0.0157	0.0021
Death	$0.7*10^{-3}$	$0.3*10^{-3}$	$1.6*10^{-3}$	$0.2*10^{-3}$

The best estimate of accident damage valuation is as follows (CEC, 1995f):

Minor accidents: 1,400 ECU
Major accidents: 94,000 ECU
Fatal accidents: 3,100,000 ECU

Based on these estimations the damage cost of public accidents is 0.016 mECU/kWh for Tunø Knob off shore wind farm and 0.018 mECU/kWh for Fjaldene wind farm.

X.3.3. Total damages related to the wind fuel cycle

The total impacts and damages which have been assessed in relation to Tunø Knob, which is an offshore wind farm are shown in Table X.18

Table X.18 Total impacts and damages in relation to Tunø Knob offshore wind farm

Impact	Quantification	Monetisation
Noise	22 summer residents, 1 farm	4 e-3 mECU/kWh
Visual amenity	Negligible	0
Atmospheric emissions	0.045 g SO_2 /kWh	0.15 mECU/kWh
	0.076 g NO_x /kWh	0.27 mECU/kWh
	0.015g CO /kWh	1 e-3 mECU/kWh
	tsp /kWh	0.01 mECU/kWh
Ozone	0.076 g NO_x /kWh	0.08 mECU/kWh
Greenhouse gases	22 g CO_2 /kWh	0.16 mECU/kWh
Public accidents	0.0074 minor accidents	
	0.0101 major accidents	16 e-3 mECU/kWh
	0.001death	
Occupational accidents	0.54 minor accidents	
	0.06 major accidents	0.022 mECU/kWh
	0 death	
Impacts on birds and shells	Negligible	0 mECU/kWh
Impacts on fish	Negligible	0 mECU/kWh
Interference with electromagnetic communication systems	0	0 mECU/kWh
Total		0.71 mECU/kWh

For the Fjaldene land-based wind farm the assessed impacts and damages are shown in Table X.19.

Table X.19 Total impacts and damages in relation to Fjaldene land-based wind farm

Impact	Quantification	Monetisation
Noise	90 houses, 30 farms	0.02 mECU/kWh
Visual amenity	7 houses	0.17 mECU/kWh
Atmospheric emissions	0.032 g SO_2 /kWh	0.11 mECU/kWh
	0.048 g NO_x /kWh	0.09 mECU/kWh
	0.009 g CO /kWh	0 mECU/kWh
	tsp /kWh	0.01 mECU/kWh
Ozone	0.048 g NO_x /kWh	0.04 mECU/kWh
Greenhouse gases	14.5 g CO_2 /kWh	0.11 mECU/kWh
Public accidents	0.0129 minor accidents	
	0.0178 major accidents	0.018 mECU/kWh
	0.0018 death	
Occupational accidents	0.97 minor accidents	
	0.11 major accidents	0.025 mECU/kWh
	0 death	
Impacts on birds and shells	Negligible	0 mECU/kWh
Total		0.60 mECU/kWh

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Title and authors

External Costs related to Power Production Technologies
ExternE National Implementation for Denmark, Appendix

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Abstract (max. 2000 characters)

The objective of the ExternE National Implementation project has been to establish a comprehensive and comparable set of data on externalities of power generation for all EU member states and Norway. The tasks include the application of the ExternE methodology to the most important fuel cycles for each country as well as to update the already existing results, to aggregate these site- and technology-specific results to more general figures.

The current report covers the detailed information concerning the ExternE methodology. Importance is attached to the computer system used in the project and the assessment of air pollution effects on health, materials and ecological effects. Also the assessment of global warming damages are described. Finally the report covers the detailed information concerning the national implementation for Denmark for an offshore wind farm and a wind farm on land, a decentralised CHP plant based on natural gas and a decentralised CHP plant based on biogas.

Descriptors INIS/EDB

AIR POLLUTION; BIOMASS; BUILDING MATERIALS; COMPUTER CALCULATIONS; DATA COVARIANCES; DENMARK; DUAL-PURPOSE POWER PLANTS; ECOSYSTEMS; ENVIRONMENTAL IMPACTS;GREENHOUSE EFFECT; HEALTH HAZARDS; METHANE; NATURAL GAS; POLLUTANTS; POWER GENERATION; SENSITIVITY ANALYSIS; SOCIO-ECONOMIC FACTORS; WIND TURBINE ARRAYS

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