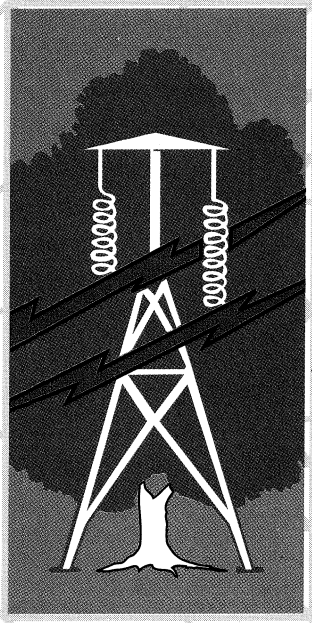
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# ExternE

# Externalities of Energy

## Vol. 9

## *Fuel Cycles for Emerging and End-Use Technologies, Transport & Waste*

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## EUROPEAN COMMISSION

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# EXTERNE

# EXTERNALITIES OF ENERGY

## Vol 9: FUEL CYCLES FOR EMERGING AND END-USE TECHNOLOGIES, TRANSPORT & WASTE

Prepared by AEA Technology, UK (editors)

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## ABSTRACT

This report describes extension of the ExternE Methodology from power generation to a range of other fuel cycles;

- End use technologies - domestic space heating and demand side options
- New technologies - integrated gasification combined cycle, fluidised bed combustion, fuel cells for stationary applications and geothermal
- Cogeneration
- Transport
- Options for waste disposal

Many complex issues arose in the course of this work. For example;

- For cogeneration there is the question of how one should allocate external costs to electrical power and heat generation.
- When considering any option for disposal of waste, external costs need necessarily to be calculated net of the externalities of alternative waste disposal options (e.g. landfill).
- For transport there is the problem of variation in externalities depending on where a vehicle is operating, speed of travel, load, and so on.

As in other reports published under the ExternE Project the emphasis is strongly on calculation of external costs rather than on resolution of specific questions such as what is the least environmentally harmful transport or waste disposal option? The main questions addressed here are whether or not it is possible to calculate externalities of the fuel cycles investigated, what is the likely magnitude of damages resulting from given technologies using defined sources of fuel, and identification of the problems encountered. Application of the results of these studies should take account of uncertainties and other issues raised in the report.

We acknowledged in our first series of reports that the results of this work are subject to significant uncertainty. However, the quantification of externalities must be looked on as starting the process of *reducing* uncertainty for decision makers. We start from a point where we know nothing about the magnitude of the impacts of different fuel cycles: we may know that impacts exist, or are likely to exist, but we do not know how large they are. In a world of limited resources it is clearly inadequate to base decisions solely on the likelihood of occurrence of an impact with no regard to the magnitude of that impact. Quantification can take us to a point where we have results describing the scale of many types of effect, an understanding of what the main sensitivities are, and knowledge of the extent of these uncertainties. It allows comparison of different technical and policy options, and the setting of priorities for environmental and other social actions. Having said this we readily concede that economic analysis on its own does not define policy because policy makers need to consider how they should react to potential uncertainties.

There is of course much fundamental research ongoing at the present time (and for the foreseeable future) that will allow refinement of the analysis presented here. For this reason we believe that it is extremely important that this work should continue. Areas where this is particularly important concern the assessment of global warming damages and impacts on human health. Substantial progress in both fields has been made in recent years, and this has been reflected in the development of the ExternE methodology.

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It should be noted that treatment of uncertainty will necessarily vary with different policy applications: an uncertainty that is of great importance in one application may be irrelevant in another. Whilst general rules for accounting for uncertainty (see Volume 7 in this series) are essential as a first step, it is therefore often useful to conduct a secondary error analysis that is customised to the case in question.

Success of the study is marked by the extensive use of the ExternE methodology by decisions makers in the European Commission, and by national and local government in Member States of the European Union. Beyond this there is also much interest in the techniques developed in the study in other parts of the world, the USA, South Africa, eastern Europe and Asia.

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Mike Holland, Dan Forster (Editors)
AEA Technology, UK.

## FOREWORD

The first series of reports from the ExternE Project was published at the end of 1995, and comprised the following volumes:

1. Summary
2. Methodology
3. Coal and lignite fuel cycles
4. Oil and gas fuel cycles
5. Nuclear fuel cycle
6. Wind and hydro fuel cycles

These reports considered the external costs of electricity generation. However, the methods developed were more widely applicable. This report describes the extension of the ExternE Project to other fuel cycles in work undertaken in the period 1996 to 1998. It is presented in three parts:

I. The fuel cycles for end use technologies, energy conservation, cogeneration and use of geothermal energy
II. External costs of transport
III. Disposal of municipal waste by incineration

Two other volumes are being published simultaneously with this: the first (volume 7 in the series) provides an update to the earlier methodology report. The second (volume 8) is concerned with quantification of the effects of global warming

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# PART 1 : EXTERNAL COSTS OF END-USE, COGENERATION AND NEW TECHNOLOGIES

# EXECUTIVE SUMMARY

## ES 1 Introduction

### ES 1.1 Background and objectives

The use of energy causes damage to a wide range of receptors, including human health, natural ecosystems, and the built environment. Such damages are referred to as external costs, as they are not reflected in the market price of energy. These externalities have been traditionally ignored.

However, there is a growing interest towards the internalisation of externalities to assist policy and decision making. Several European and international organisms have expressed their interest in this issue, as may be seen in the 5th Environmental Action Programme, in the White Paper on Growth, competitiveness and employment, or the White Paper on Energy, all from the European Commission. This interest has led to the development of internationally agreed tools for the evaluation of externalities, and to its application to different energy sources.

Under Joule III, this project has been continued with three distinguished major tasks: ExternE Core for the further development and updating of the methodology, ExternE National Implementation to create an EU-wide data set and ExternE-Transport for the application of the ExternE methodology to energy related impacts from transport. The current report is part of the ExternE Core project.

The objective of this sub-task of the ExternE Core project is to apply the existing framework to new domains, as there are:

- end-use technologies:
  - domestic space heating with oil, natural gas and electricity, and
  - demand-side options (here with respect to refrigerators, windows and heat exchangers);
- cogeneration; and
- future technologies (here IGCC, fluidised bed combustion, fuel cells for stationary applications and use of geothermal energy).

Thereby, new methodological issues have to be addressed in some cases and in others new priority impact pathways have to be analysed, thus extending the existing accounting framework.

The output of the project could be very useful for policy-making, both at the national and EU level as the results obtained provide a good basis to start the study of the internalisation of the external costs of energy, which was frequently cited as one of the objectives of EU energy policy. Other possibility is to use the results for comparative purposes. The site sensitivity of

the externalities might encourage the application of the methodology for the optimisation of site selection processes, or for cost-benefit analysis of the introduction of cleaner technologies.

The present report is to be seen as part of a larger set of ExternE publications. It summarises the results of the sub-task on the 'Extension of the Accounting Framework' of the EXTERNE Core project.

## ES 2 Methodology

The methodology used for the assessment of the externalities of the fuel cycles selected has been the one developed within the EXTERNE Project. It is a bottom-up methodology, with a site-specific approach, that is, it considers the effect of an additional fuel cycle, located in a specific place.

To allow comparison to be made between different fuel cycles, it is necessary to observe the following principles:

- Transparency, to show precisely how the work was done, the uncertainty associated to the results, and the extent to which the external cost of any fuel cycle have been fully quantified.
- Consistency, with respect to the boundaries placed on the system in question, to allow valid comparison to be made between different fuel cycles and different types of impact within a fuel cycle.
- Comprehensiveness, to consider all burdens and impacts of a fuel cycle, even though many may be not investigated in detail. For those analysed in detail, it is important that the assessment is not arbitrarily truncated.
- These characteristics should be present along the stages of the methodology, namely: site and technology characterisation, identification of burdens and impacts, prioritisation of impacts, quantification, and economic valuation.
- Quantification of impacts is achieved through the damage function, or impact pathway approach. This is a series of logical steps, which trace the impact from the activity that creates it to the damage it produces, independently for each impact and activity considered, as required by the marginal approach.
- The underlying principle for the economic valuation is to obtain the willingness to pay of the affected individuals to avoid a negative impact, or the willingness to accept the opposite. Several methods are available for this, which will be adopted depending on the case.

## ES 3 New Methodological Issues

### ES 3.1 Allocation of External Costs between Electricity and Heat

The allocation of costs to either heat or electricity is a well-known problem in the field of internal cost accounting of cogeneration plants. Thus, a number of alternative approaches have been proposed and used in the past. Here, three approaches are examined:

1. *Crediting for heat and electricity* is a commonly used and accepted method for the calculation of internal costs. However, as consistency is impaired when avoided costs due to substituting other technologies are included in the damage costs calculation, the approach is judged to be not appropriate for the EXTERNE accounting framework.
2. *Allocation based on energy or exergy content of co-products* allows a clearly-defined split-up of the damage costs according to the cogeneration plant's characteristic and does not depend on the context. Thereby, the product's exergy indicates the higher 'value' of mechanical or electrical energy. In any case however, the results are difficult to interpret with regard to energy/economy problems.
3. *Allocation based on the final products' price*: prices are – assuming a perfect market – probably the best indicators for the 'utility' of the products. However, the prices of electricity and heat are highly dependent on the customers demand characteristics and other non-technical parameters.

While the first approach does not appear to be appropriate for the EXTERNE accounting framework, a parallel use of the second and third allocation rules is recommended.

### ES 3.2 Analysis of Demand-Side Options

Damage costs of demand-side options basically are due to production, installation and disposal. These costs can be related to a certain amount of energy that can be saved by initially introducing these technologies or by replacing standard/conventional technologies by advanced energy efficiency technologies. In the case of replacing less energy efficient technologies with advanced new devices, the incremental impact of the new technology compared to the conventional technology is the basis for the damage cost calculations.

The burdens of the upstream (pre-production) processes are calculated with EMI 2.0 – an input-output model that has been extended by sector-specific emission-coefficients for the German economy split up into 58 production and service sectors.

## ES 4 Reference Technologies, Sites and Emissions

### ES 4.1 Domestic space heating

In the past many studies comparing energy use and emissions of different domestic space heating options used 'final energy' as functional unit, because final energy, e.g. the oil or gas

supplied to the different households, is statistically recorded, the produced heat is not. However, such an approach neglects that different heating options employ different amounts of final energy to satisfy a specific heating task. Therefore, it is more appropriate to define a certain heating task and to take the quantity 'useful energy produced' as functional unit, as the more recent studies have done. Here, the selected heating task is the central space heating of a one-family house. It is assumed that this house is located in Lauffen, 35 km north of Stuttgart. The specified heating task can be met by boilers with a nominal capacity of 20 kW. Technologies of the present state of technology are selected:

- oil: a low-temperature boiler with an atomising oil burner (blue-burner); annual utilisation ratio (incl. distribution losses) 88.1%;
- gas: a condensing boiler; annual utilisation ratio (incl. distribution losses) 100.9%; and
- electric: decentral thermal storage heater supplied by 'night current'; annual utilisation ratio (incl. distribution losses) 98.5%.

For the electric heating systems essentially all emissions are due to the supply of electricity to the household, of course. Thereby, the emission quantities depend on the composition of the electricity supply mix for thermal storage heating. Data for this mix are available from the eight largest German electric supply companies, which show a very large variation, from a share of nuclear energy of 82% to a share of lignite of 98%.

### ES 4.2 Demand-side options

One of the biggest existing potentials for demand-side options can be found in the area of private households. For example about 12% of the total electricity demand of private households (excluding electric heating of air and water) is used for refrigeration purposes. As the most energy-efficient refrigerators consume only about 2/3 of the energy demanded by standard refrigerators, a higher energy-requirement in the production of energy-efficient refrigerators can be positively balanced by the energy-savings during the lifetime. Another possibility is to reduce the energy needed for domestic space heating by reducing the heat losses via the windows. Accordingly, two technologies are analysed here:

- refrigerators with and without evaporation compartment: difference between average and present technology, and
- windows: difference between window with two panes to windows with three panes.

As for the private households, there are demand-side options to reduce the energy demand of the manufacturing sector. Here, heat exchangers are examined, treating them as a new technology replacing an old one. Hence, the total impacts of production, installation and disposal are the basis for damage costs accounting.

### ES 4.3 Cogeneration

Combined heat and power (CHP) plants in three different countries are analysed. For Germany two reference technologies are considered:

- a small-sized combined cycle gas-fired CHP plant, and
- a large coal-fired CHP plant, equipped with dust precipitators, FGD and DENOX facilities.

The CHP plants are located in Mannheim and Stuttgart, respectively, where district heat networks are available.

The Italian case study analysed the damage costs of an industrial CHP plant. The reference technology is a typical example of those under construction in the nineties in Italy and in Europe where independent generation is allowed. The medium-sized plant has been actually built in an existing chemical plant that produces basic chemical products in a simple, but energy-intensive process.

Two case studies are considered for Finland. The first, a wood-fired CHP plant, which is comparatively small, is a relatively simple basic case of coproduction of district heat and electricity. The second is a more complicated plant, which supplies in addition to district heat and electricity *and* steam to a nearby industrial plant. It is a medium-sized CHP plant fired by peat and biomass. Both cases represent actual plants currently operated in Finland near Forssa and near Jyväskylä.

### ES 4.4 IGCC technology

A power plant using an Integrated Gasification Combined Cycle (IGCC) technology has been built in Puertollano, 230 km south from Madrid (Spain), by Elcogas, a consortium formed by several European utilities and engineering companies. This specific IGCC technology is one of today's most promising electricity generation options, both from the environmental and the efficiency point of view. It will allow an efficient and environmentally-friendly use of national coal, and also of a refinery residue, petroleum coke. When operating, the plant will be the world's largest IGCC plant.

### ES 4.5 FBC technology

Due to environmental considerations and due to the fuel flexibility of fluidised bed combustion (FBC) this technology has been intensively investigated and developed in the last three decades. Here, two pressurised fluidised bed combustion (PFBC) technologies have been analysed, assuming for both a site at Lauffen, 35 km north of Stuttgart:

- a coal-fired combined cycle power plant with PFBC of the present state of technology, and
- a coal-fired combined cycle power plant with PFBC *and* additional gas firing as predicted for 2005.

Fluidised bed coal combustion is known to produce much higher $N_2O$ emissions than conventional coal-fired units. $N_2O$ levels of about 100–250 mg/Nm$^3$, under unfavourable conditions even up to 500 mg/Nm$^3$, are given in the literature. This compares with typical levels of $NO_x$ in the range of 200–400 mg/Nm$^3$ from fluidised beds without secondary emission reduction measures. Here, $N_2O$ emission levels have been varied between 100–250 mg/Nm$^3$ to analyse the influence on the results for global warming.

### ES 4.6 Stationary application of fuel cells

Unlike conventional thermal engines, fuel cells generate direct current electricity and heat by combining fuel and oxygen in an electrochemical reaction. Hence, fuel cell efficiencies are not limited by the Carnot factor as with the (conventional) Rankine cycle technologies because there is no need for an intermediate combustion step and boiling of water. Most fuel cells are still in the development stage. Potential markets are the power sector, combined heat and power (CHP) systems and transportation. The objective in this study is the analysis of stationary applications of fuel cells. Two cases are analysed:

- a phosphoric acid fuel cell (PAFC) CHP plant, as PAFC is the fuel cell with the highest development stage, and
- a molten carbonate fuel cell (MCFC) CHP plant. Momentarily, a European syndicate works on the commercialisation of the MCFC.

Both fuel cell CHP plants are gas-fired and are located near Stuttgart. The $SO_2$, $NO_x$ and particulates emissions of the fuel cell CHP plants are very small or even zero. With the exception of $CO_2$ – the emissions of the reference energy systems are dominated by the gas fuel cycle *and* the production of the CHP plant. The quantification of the emissions from the upstream processes shows that for both fuel cell CHP plants the production of the fuel cell contributes considerable to the $SO_2$, $NO_x$ and particulates emissions of the whole reference energy system. This is caused by the energy-intensive processes to produce the platinum and nickel used in the fuel cell stack.

### ES 4.7 Geothermal-to-electricity fuel cycle

The analysis focussed on a geothermal power plant located in São Miguel island in the North slope of Água de Pau, between Cachaços and Lombadas, part of the municipality of Ribeira Grande and the district of Ponat Delgada. The reference plant produces about 80 000 MWh annually, which represents about half of the total electricity consumption of the island.

The main technological modules are: (i) the plant with 5 energy converters, (ii) 4 geothermal wells with about 1 500 m depth each and complementary units (pipelines, steam separator and collector) and (iii) energy transmission network.

The flux of $CO_2$ from deep geothermal structures is associated with the metamorphism of underground carbonate reservoirs, and namely with the metamorphism of marine carbonate rocks. However, there is a considerable variation in the amount of $CO_2$ emitted from different geothermal wells. The lowest $CO_2$ emissions (13 g/kWh electricity produced) are recorded for the geothermal well in Wairakei, New Zealand, the highest with 804 g/kWh for the geothermal well under analysis here. These high emissions could be due to the low efficiency of the power plant here and/or the high rate of metamorphism of the subterranean carbonate reservoirs. The last one seems to be more important.

## ES 5 New Priority Impact Pathways

The technologies under analysis in this report in most cases use fossil fuels, therefore new priority impact pathways are neither caused by their generation stage nor by their upstream processes. The exception is the geothermal-to-electricity fuel cycle, which is so completely differently structured that on the one hand many priority impact pathways analysed so far are not relevant and on the other hand new impacts had to be looked at. Four potential new priority impacts have been identified and analysed, accordingly.

### ES 5.1 Potential impacts due to the locations of a geothermal plant in a zone of seismic and volcanic activity

In case of a seismic or volcanic event it is possible that the well is fractured and geothermal fluid contaminates the upper aquifers making its water improper for drinking or for other valuable purposes. The impact on the aquifers is temporary as with time the channels opened during the seismic event would be obstructed by scaling. Damages that may be caused to the facility during a seismic event are considered as internalised: the plant owner has incorporated such risk into the economic evaluation of the project, e.g. by an insurance.

For the time drinking water could not be taken from the aquifers, water had to be transported from other regions into the island. The additional costs to provide drinking water has been quantified by assessing the probability that an earthquake exceeding a certain magnitude occurs and that well ruptures caused by the earthquake affect the underground water sources.

### ES 5.2 Impacts on water quality from heat mining

During the drilling of the wells and the testing of the productive ones, underground waters that are normally used for farming, breeding and for other domestic or agricultural purposes can be affected and become improper for use. Again, the impact is temporary. It is assessed analogous to the approach used for the well ruptures during seismic events.

### ES 5.3 Environmental impacts from geothermal fluid release

Impacts of heat mining and waste disposal on surface (fresh and coastal waters) and groundwater quality can be significant. Contaminants such as boron, mercury and arsenic are often present in geothermal waters. When this water is discharged into water courses, the contaminated downstream waters can become polluted and thus be harmful to farming, fisheries, and other water supply purposes. Thermal pollution can result from the direct disposal of insufficiently cooled brine. Although in some cases an increase of water temperature can be beneficial, e.g. increasing biological production, in others, increases in water temperature can have negative impacts. As neither original data nor adequate damage functions are available for these impacts, only a qualitative assessment can be carried out.

### ES 5.4 Impacts from hydrogen sulphide on human health and local amenity

Geothermal fluids almost invariably contain some amount of hydrogen sulphide ($H_2S$), which even in low concentrations has an unpleasant smell ('bad eggs'). When strongly concentrated, it paralyses the olfactory nerves and thus cannot be smelled anymore. Being heavier than air, it tends to collect in low-lying pockets. Fatalities are known to have occurred in the vicinity of fumaroles, but no case has so far been associated with geothermal exploration or power production.

The potential concentrations of $H_2S$ around the well are calculated and compared to several threshold values for different health effects. From this analysis it was concluded that the odour will be detected by the human nose but the concentrations do not reach the level to be registered as 'bad egg' smell or even to cause physiological symptoms.

## ES 6 Overview of the results

Without global warming, in nearly all analysed fuel cycles the mortality effects – especially due to sulphate and nitrate aerosols – dominate the results. For global warming the results produced by the global warming task group of the EXTERNE Core project have been used. The task group suggested two ranges, a 95% confidence interval based on the Monte-Carlo-Simulation of the climate cost model used and an illustrative restricted range derived from the base case estimates for discount rates between 3 and 1%. In general, the global warming results are of the same order of magnitude as all other quantified damage costs together.

How the choice of the allocation affects the quantified damage costs for electricity and heat has been analysed for the two German CHP plants and the Italian CHP plant. Figure ES.1 shows the respective results for electricity. While the electricity damage costs of the Italian rules do hardly differ between allocation based on exergy and allocation based on price, for the two German CHP plants the external costs of electricity are slightly higher if allocated based on exergy as if allocated based on price. The developed approach provide a framework to calculate external costs from a CHP plant per unit electricity or heat produced consistent with the overall EXTERNE approach.

The quantified results for the demand-side options under analysis in this report (Figure ES.2) in comparison to the damage costs quantified for heat (Figure ES.3) and electricity (Figure ES.4) demonstrate the advantages of such measures again. However, the results for the different options also show that for a cost-efficient approach it should be analysed carefully, which option to support first.

For the heating systems for central heating of a one-family house analysed in this report damage costs of about 1–4 mECU per MJ useful energy produced (excluding global warming) are quantified. Including global warming the results increase considerably, especially for the electric thermal storage heating system (see Figure ES.3). However, for electric heating the results vary substantially depending on the electric supply mix. Not surprisingly, the higher

the share of nuclear energy in the supply mix is, the lower the quantified damage costs are. Compared to the results for the German coal CHP plant the results for the gas condensing boiler are quite high. However, thereby should be taken into account that most of the damages caused by the CHP plant are allocated to the electricity produced.

Due to the characteristics of the generation technology total impacts from the IGCC fuel cycle are small compared to a conventional Spanish coal power plant. Within the upstream stages of the fuel cycle, the oil refining and coke production stage deserve special attention. This stage causes the highest $SO_2$ emissions of the whole fuel cycle, due to the high sulphur content of the oil used for coke production.

In this report the highest damage costs have been quantified for the two PFBC power plants. Without secondary emission reduction measures, the $SO_2$ and $NO_x$ emissions of these plants are higher than those of modern plants equipped with a DENOX facility. There is some concern because of the comparatively high $N_2O$ emissions of this technology. The damage costs quantified for these $N_2O$ emissions are significant but small if compared to the damage costs quantified for the $CO_2$ emitted.

Fuel cells have the potential to contribute in a significant way to emission reduction, especially of $CO_2$ and $NO_x$. With the exception of global warming, the damage costs of the generation stage are negligible compared to the annual damages of the upstream gas fuel cycle and of the fuel cell production.

The damage costs quantified for the geothermal power plant under analysis here are dominated by global warming because of the unusually high $CO_2$ emission factor of this plant. Other important effects of this fuel cycle are impacts on water quality. While some of them could be quantified, water quality impacts due to heat mining, which are considered to be significant, could not be assessed.

## ES 7 Conclusions

External cost estimates are calculated following a standardised methodology that is widely accepted now on the international level, and similar data sets are produced in all member states of the European Union. Although there are significant remaining uncertainties in some areas, results indicate that external costs of some fuel cycles are high enough to affect energy policy decisions. The work carried out for this report successfully extended the methodology where needed. In summary, the analysis proved that the methodology developed so far in the EXTERNE project can be applied to most fuel cycles and applications.

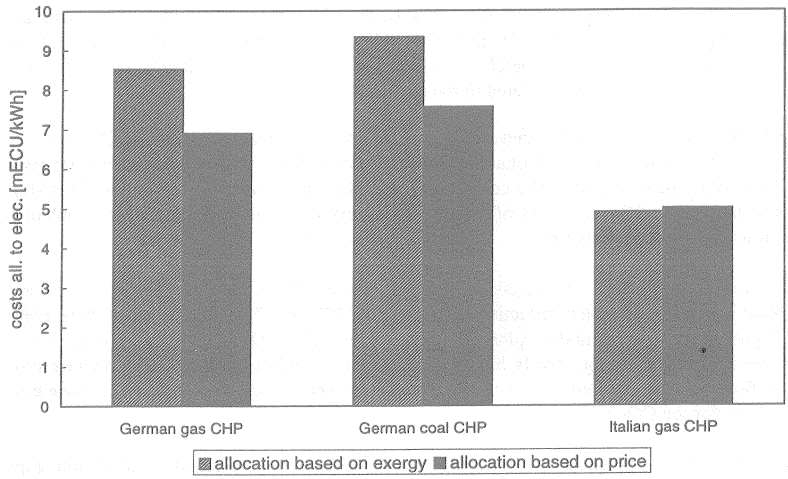



**Figure ES.1** Comparison of allocation based on exergy to allocation based on price – Damage costs (without global warming) [mECU/kWh] quantified for the German and Italian cogeneration plants and allocated to electricity

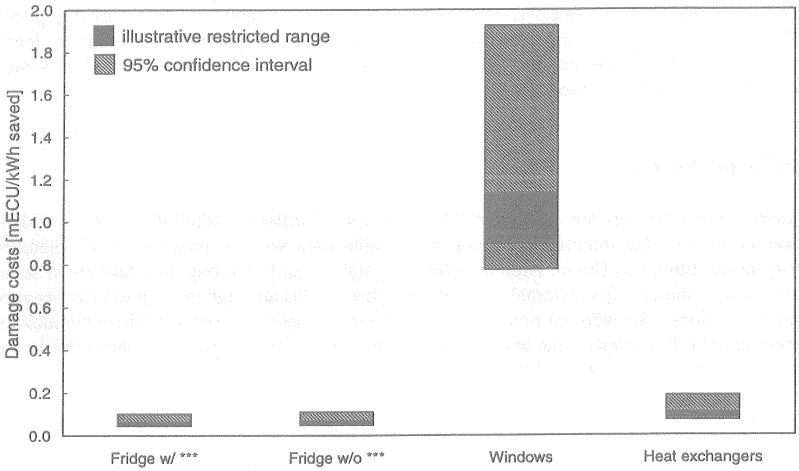



**Figure ES.2** Damage costs [mECU/kWh] including global warming quantified for the demand-side options analysed in this report

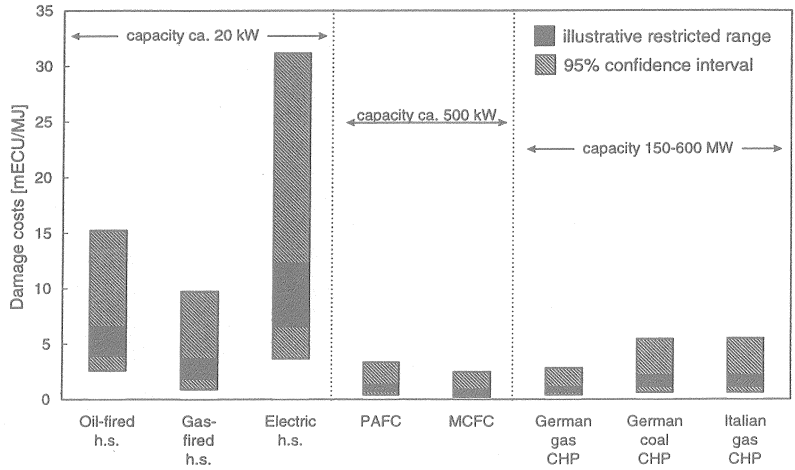



**Figure ES.3** Damage costs [mECU/kWh] including global warming quantified for the heat producing technologies analysed in this report (damage costs for CHP plants allocated to electricity based on exergy)

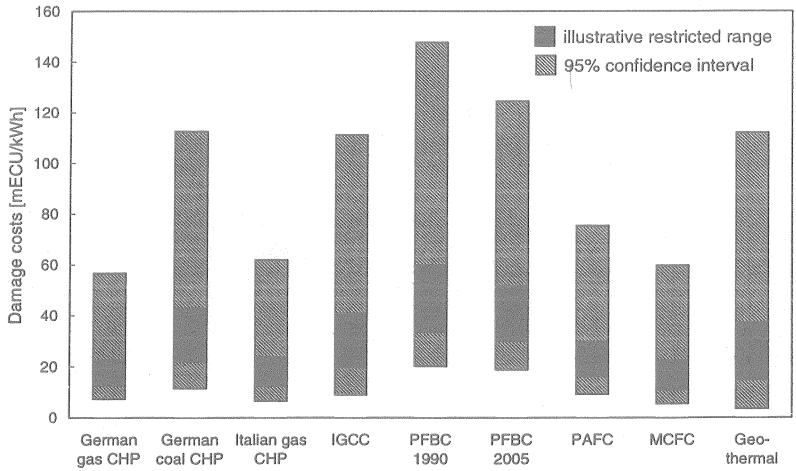
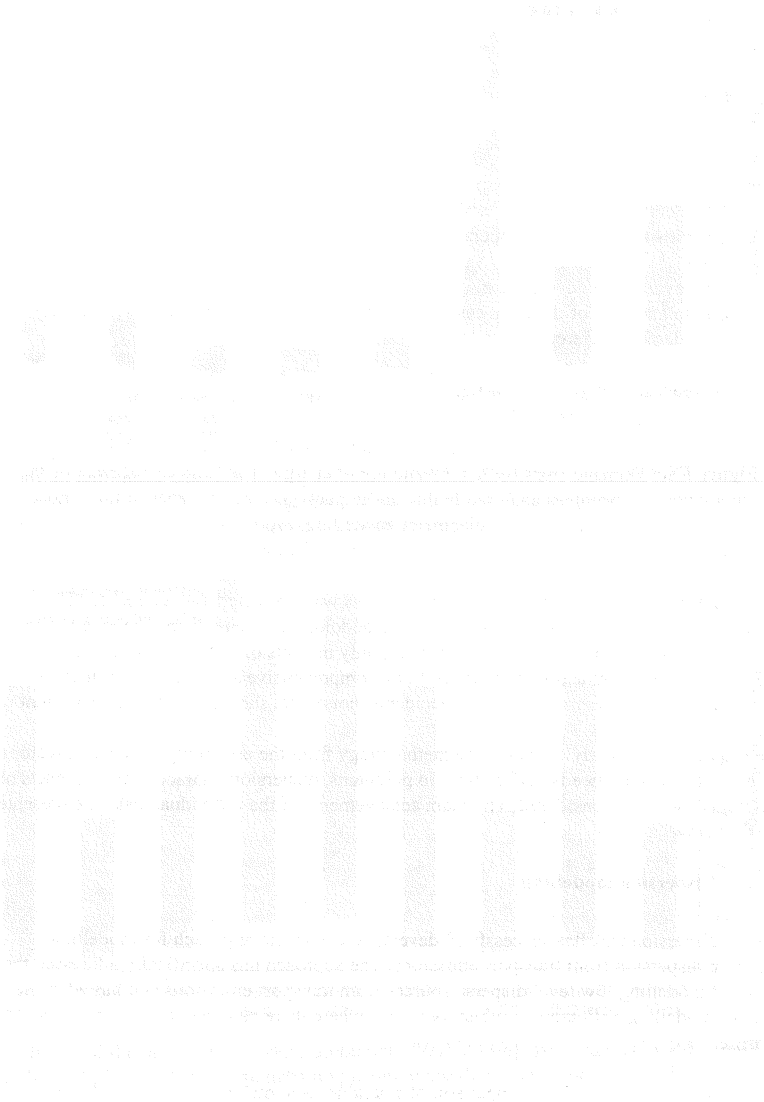



**Figure ES.4** Damage costs [mECU/kWh] including global warming quantified for the electricity producing technologies analysed in this report (damage costs for CHP plants allocated to electricity based on exergy)



# PART 2 : EXTERNAL COSTS OF TRANSPORT

# EXECUTIVE SUMMARY

## ES 1 Introduction

The aim of this project was to develop a new methodology for quantifying energy related environmental externalities of transport based on a bottom-up approach. This methodology allows the calculation of marginal external costs of different transport activities in a detailed and consistent way, considering operation and the life cycle of the transport means.

This report reviews the developed methodology and presents the major results obtained in its application to a variety of different case studies, assessing passenger and goods transport with different modes and different technologies.

### ES 1.1 Impact pathway methodology

The project adopted the 'impact pathway' approach for the assessment of impacts and associated costs resulting from the supply and use of energy. The 'impact pathway' is the sequence of events which links a 'burden' to an 'impact'. The main stages of the 'impact pathway' are: emission modelling, dispersion modelling, estimation of physical impacts using exposure-response functions and finally the valuation of impacts.

A considerable number of priority impact pathways was identified and addressed. The focus lay on the development and application of a methodology for quantifying the external costs due to energy conversion processes. Therefore, only impacts of airborne pollutants and greenhouse gases were assessed in detail. For a comprehensive assessment of transport externalities, of course, impacts due to accidents, noise, etc. should be taken into account.

The project teams jointly extended the methodology from the electricity generation sector to the transport specific needs. Differences in pollutants, dispersion characteristics, impacts of pollutants had to be considered. The main achievements of the individual tasks are presented briefly below.

### ES 1.2 Dispersion modelling

The air dispersion task has successfully developed a working approach for modelling pollution dispersion from transport emissions. The approach has specifically addressed the issues of modelling low level disperse sources from transport emissions and has addressed the issue of modelling pollutants in urban locations, where there can be extremely high receptor densities.

On the basis of a review and comparison of available local dispersion models, the project has selected and demonstrated the ROADPOL model for quantifying local-scale pollutants.

ROADPOL is a Gaussian dispersion model which estimates pollutant concentrations for line sources.

In addition, the modelling task has updated the regional assessment of transport pollutants, extending the assessment to include carbon monoxide and VOC dispersion at a regional level, and differentiating between the particulate size distribution for transport and power station emissions (so as to properly evaluate the finer nature and longer range of particulate emissions from transport).

### ES 1.3 Health effects

For the health effects of transport related pollutants, the epidemiological literature was reviewed, concerning the 'classical' pollutants (particulate matter, PM; ozone, $SO_2$, $NO_2$, CO) and expanded to transport specific pollutants (especially carcinogens).

Epidemiological literature was surveyed in the context of other evidence to form a view on what ambient air pollutants are shown as associated with adverse health effects (acute or chronic), and for what specific endpoints; which of these associations may reasonably be interpreted as *causal*; what studies provide a basis for a good set of exposure-response (E-R) functions; and how if at all should the E-R functions from individual studies be adapted for use in ExternE.

E-R functions are considered for acute mortality, respiratory hospital admissions, cerebrovascular hospital admissions, cardiovascular hospital admissions, emergency room visits, restricted activity days, acute effects in asthmatics, respiratory symptoms in the general population, chronic mortality, and chronic morbidity.

For mortality impacts, the basic inputs are the exposure-response function from /Pope et al. 1995/; the age-specific death rates from the baseline population in whom effects are being estimated and the age structure of the baseline population. With this information, life table calculations were carried out to calculate the years of life lost (YOLL), attributable to the pollution increment.

For carcinogens, the unit risk approach was followed for the pollutants benzene, benzo[a]pyrene, 1,3 butadiene, and diesel particles, along with formaldehyde and ethylene oxide for sensitivity analyses only.

### ES 1.4 Impacts on natural and man made environment

Besides impacts on health, emissions of air pollutants from transport activities impose burdens to vegetation, ecosystems and materials. Due to complex interactions, effects to individual species are often unknown and causality of effects are difficult to prove.

In the case studies, material degradation due to $SO_2$ and wet acid deposition on zinc, galvanised steel, sandstone, natural stone, limestone, mortar, rendering and paint is quantified. For materials soiling by primary particles an approach based on /Pons et al. 1995/ was applied, linking cleaning costs for buildings to the particle concentration increment. Concerning effects on crops due to $SO_2$, rye, oats, potatoes, sugar beet, wheat, barley were

considered. Crop losses due to ozone were taken into account in the damage factors for ozone damages.

Although new effects were reviewed from latest research results, e.g. visibility impairments, some of them could not be implemented properly within the current phase of the Project due to a lack of necessary environmental models and data.

From the results it can be concluded that effects from these pathways have only minor influence on the total external costs because of the much higher damages from health effects. However it has to kept in mind that the impact assessment on the natural environment is far from being complete in particular for ecosystems.

### ES 1.5 Global warming

Damages due to climate change are one of the most important categories of fossil fuel emission related damages, but also amongst the most uncertain and controversial. The impacts of incremental climate change due to emissions from the European transport sector are truly global in range. Moreover, they will extend over many generations and a very wide range of resources and human activities.

The Inter-Governmental Panel on Climate Change (IPCC) has undertaken detailed analyses of the science of climate change, the physical impacts and potential responses and the socio-economic dimensions. The work under way from the ExternE Core Project, extending the work of IPCC, could not be finished in time for the use in the transport project. However, preliminary data from the ExternE Core Project indicated results broadly consistent with the IPCC range, if catastrophic and higher order damages are neglected. In the absence of ExternE values, it was recommended that the damage range quoted by the IPCC should be used.

### ES 1.6 Up- and downstream processes

One of the central features of the ExternE Methodology is that the impacts from all stages of the fuel cycle are considered. Thus, for transport the impacts from fuel production, vehicle production, maintenance and disposal, and the supply of infrastructure were considered in addition to vehicle use.

Because a detailed collection of primary data from up- and downstream processes was beyond the scope of the project, estimates had to be based on existing studies. These studies give average values for energy use and emissions of the different life-cycle stages. The nature of relevant emissions were carefully checked and adjusting estimates were made where necessary for a correct impact assessment. Damage factors per tonne of pollutant emitted which can be derived from impact assessment on the European scale, were applied on the emissions to calculate damages.

### ES 1.7 Economic valuation

Economic valuation of damages in the Project followed the approach developed in the previous phase of the ExternE project. In addition, transport specific endpoints had to be

considered and the respective monetary values had to be derived. Furthermore some of the economic values used in the previous phases of ExternE had to be adjusted based on a new literature survey and new insights in mortality valuation. All values are expressed in 1995-prices.

Major changes in the magnitude of results for mortality impacts were introduced by adopting the YOLL (Years of Life Lost) approach instead of valuing the 'case' of death. For this the VLYL (Value of a Life Year Lost) is derived from VSL (Value of a Statistical life), to obtain WTP per year of life.

## ES 2 Results of the case studies

The methodology developed was applied to a series of case studies in Germany, the United Kingdom, the Netherlands, France, Greece and Italy. Environmental externalities due to passenger and goods transport were assessed for road, rail and inland waterway transport technologies. Different fuels and emission reduction technologies were studied.

It can be concluded that health impacts dominate the damages quantified in this study; in particular, mortality due to primary ($PM_{2.5}$) and secondary particulates (nitrates, sulphates). Carcinogens, which were expected to play an important role due to their high specific toxicity, proved to be of much lower importance compared to the particles.

It was found, that the population density around a road is a key parameter for the magnitude of impacts, particularly for diesel fuelled vehicles. This effect is caused mainly by the importance of the primary particles for the total damage.

Location dependency is less important for petrol cars, as they emit much less primary particles and the results are therefore dominated by the secondary pollutants, which form on the regional scale.

A comparison of the different technologies and modes shows, that due to their high primary particulate emissions, diesel vehicles cause higher quantifiable impacts than comparable petrol, LPG, CNG, biomethanol and bioethanol vehicles. It can be observed that in tendency, cars cause higher damages than trains per passenger kilometre, even if very 'clean' petrol cars were taken. Of course, the average occupancy rate is the crucial parameter for this comparison.

The priority impact pathways were reviewed for aircraft transport as well. Due to substantial differences in dispersion of the airborne pollutants emitted in high altitudes it was not possible to provide estimates for the full range of pathways.

# PART 3 : EXTERNALITIES OF WASTE

## EXECUTIVE SUMMARY

The present study has been prepared within the Joule II and III programmes as part of the "ExternE Project", a major research project promoted by Directorate-General XII of the European Commission, aimed at the estimation of monetary values of external costs entailed by several fuel cycles. The project has been carried out by over 40 European scientific institutions working on common methodological grounds in order to ensure the largest consistency and international comparability of results. It follows the "impact pathways" approach which envisages the incorporation of current scientific knowledge of impacts arising out of fuel cycles externalities and applies common valuation criteria to all of them. The methodology has been outlined in other ExternE volumes (EC DG XII 1995a, 1995b, 1995c and 1998), and it is not further described here.

### ES 1 Reference Technologies, Locations and External Cost Accountancy Issues

Energy may be recovered or saved by several practices in the field of solid waste management: recovery of biogas generated by landfill sites, incineration, combustion of waste-derived synthetic fuels, reuse or recycling of energy-rich waste fractions. The present study is limited to municipal solid waste (MSW) incineration (mass burning) but this does not involve it being the most effective or efficient energy management practice in the field. It is only the technology which currently ensures the largest energy recovery from MSW within advanced countries.

The study analyses impacts from a MSW incineration plant with electric energy recovery. The specific plant analysed has not been built yet, but will be located in the west outskirts of Milan. It is planned to have an average MSW disposal capacity of 900 tonnes/day. With a gross calorific value of refuse around 15000 kJ/kg and a net thermal efficiency of 21%, this involves an electricity generating capacity of 33.3 MW, and a net electricity production of 888 kWh/tonne (with a load factor of 85%).

If MSW incineration is regarded as a fuel cycle, account must be taken that its adoption actually displaces other waste disposal solutions which may entail in turn externalities and must therefore be evaluated and subtracted from those of incineration. It is assumed that incineration actually displaces landfilling of MSW, the most popular waste disposal practice in the world.

The avoided disposal plant is assumed to be a modern, sanitary landfill equipped with biogas recovery and energy generation. In particular, we considered the plant of Cerro Maggiore, very recently closed, whose capacity was about 2000 t/day. The landfill generating capacity per tonne of waste disposed is much lower as compared to the incinerator, and follows a different time path due to delays in landfill gas production (of up to 15 years) from each MSW unit.

Thus, it is assumed that the landfill gas-fired energy recovery engines, with a net thermal yield of 11.7%, generate 48 kWh/tonne (MSW) distributed over a much longer time span, so that average electricity capacity is 4.0 MW and peak capacity is about 11.5 MW.

External costs of MSW incineration must therefore be netted of those of the landfill site and the result should be attributed to its electricity generation less what is foregone by avoiding the MSW landfill.

Furthermore, externalities which are not sensibly modified by turning from MSW landfilling to incineration should not be included. This point would apply to MSW transportation if the location of plants is supposed to be the same, and to most soil and groundwater pollution as contaminants are concentrated in bottom ashes, slag and dust electrostatic precipitators which must be eventually landfilled anyway (assuming that the leachability of these contaminants is little affected by the treatments given). These impacts, however large they may be, are therefore approximately balanced by those of the MSW landfill, and not analysed further.

The only exception considered for its practical relevance is the substitution of an urban incinerator for a rural landfill, which saves some transport external costs.

The analysis is performed for two real sites/plants, even if, for different reasons, they are actually not operating. The site identified for the incinerator is close to via Silla, a street in the western outskirts of Milan, which is a very densely populated area. The incinerator is still to be built. As regards the landfill, the study considers the plant placed close to Cerro Maggiore, a small town about 25 km North-West of Milan. Although the landfill site is sometimes referred as "rural" in the report, the area surrounding this plant has a rather high population density, close to an average of 2000 inhabitants per square km.

## ES 2 Impact Assessment

Table 1 and Table 2 below summarise priority impact pathways and the state of advancement for each of them. Most of the impacts were analysed according to the set of Exposure-Response Functions (ERF) and monetary values identified within the ExternE programme. Only a few of them, notably those related to perceived (visual and odour) impact of the landfill site and the incinerator, were quantified with a specific hedonic pricing study.

The analysis of impacts of atmospheric emissions was performed using Ecosense 2.0 (Krewitt W., 1997), for both the *local* (100 x 100 km squares centred on the plant) and *regional* (European) range. A different Gaussian plume model was used to model the local dispersion of odours originated by the landfill, as very detailed data were required to implement the hedonic pricing technique.

MSW transport external costs were assessed according to the ExternE – Transport methodology (EC DG XII, 1997), which involves the use of ROADPOL to estimate atmospheric dispersions from mobile sources.

ERF were chosen - when available - among those used for other ExternE fuel cycle studies, particularly those regarding coal which is characterised by most similar impacts. Some appropriate ERF and threshold limit values were taken from specific literature as regards cadmium, mercury, lead and chlorinated organic micropollutants emitted by incinerators, and for organic carcinogenic components of landfill gas (benzene). Heavy metals and Cl-organic micropollutants were treated as persistent and likely to cause adverse health effects by indirect pathways as well as by direct inhalation.

Morbidity effects are attributed to particulates, nitrogen and sulphur oxides, and some micropollutants. A previous study (IEFE 1995) proved that hydrogen chloride and fluoride emissions are likely to cause concentrations below danger thresholds.

Accidents in the plant construction stage and in MSW road transport are estimated by scaling statistics related to investment costs and to road accidents.

Greenhouse effects which are particularly important for this fuel cycle are analysed by considering the specific time path of emissions and different atmospheric decay times of each relevant gas (carbon dioxide and methane).

## ES 3 Economic Valuation and Results

As for the other stages of the impact pathway, all the effects are evaluated in monetary units as far as possible according to the methodology developed within the ExternE project (EC DG XII 1995b). All valuations follow therefore the willingness to pay approach and should be comparable with other ExternE studies (*ibidem*).

Two sets of impact pathways are worth being mentioned because of their peculiarity with respect to the other technologies analysed in the ExternE programme. The first one is given by cancer cases due to persistent (organic and inorganic) micropollutants. The evaluation of this pathway requires the description and consideration of all different (direct and indirect) pathways that a pollutant can follow before affecting human health (and other possible targets). The second impact is represented by the "amenity losses" due to the visual impact and to the public perception of odours and traffic congestion close to a MSW treatment plant (and in particular to a landfill). Both aspects are analysed and quantified in this report.

Agricultural impacts are evaluated at international prices.

The 3% discount rate is used as best estimate throughout the whole text where not otherwise specified. Global warming impacts are assessed using a range of monetary values per mass unit of greenhouse gases emitted derived from other original modelling work undertaken within the externE programme.

Burdens and impacts analysed are given in the last two columns of Table 1 for incineration and Table 2 for landfill sites. The results are summarised in Figure 1 in terms of ECU per tonne (MSW), and further specified in more details in the report.

## ES 4 Warning on Missing Impacts and Utilisation of the Study

It must be recalled that all the results quantified by this study are bound to specific assumptions and, moreover, they are highly related to the plant locations considered. Transfer of our estimates to other sites should carefully consider their characteristics as regards in particular, technologies emission rates, meteorology, use of land, and population density of affected areas.

In addition, it should be recalled that the present study could not estimate some impact pathways. No evaluation was made of soil and groundwater pollution by landfill sites. This may not be very relevant for modern, isolated plants where leakage is contained under proper management conditions, but little can be said in case of improper use, e.g. by landfilling of hazardous chemicals.

The lack of a local, reliable database for the evaluation of materials damages should not lead to large biases for the estimation of externalities of incineration and landfilling.

Another missing item is given by disamenity impact of the incinerator as no specific study could be undertaken. An hedonic pricing study on a plant located about 30 km from the incinerator considered by our study suggested that the amenity losses due to this technology may be negligible (see section 3.6). It is however difficult, and often not sensible to transfer these results from site to site. The only plausible conclusion that might be drawn from these studies is that the disamenity impact caused by incinerators is expected to be lower as compared to the landfill. More precise statements would need to be supported by further research.

Utilisation of results for planning purposes or setting levies or subsidies to waste disposal technologies should seriously take into account the limitations mentioned above.

Finally, the reader should keep in mind that besides waste incineration and landfilling there are other major practices for waste treatment - recycling above all - that could lead to important energy savings, but were not addressed within the present study. General conclusion on the economic rationality - from a social welfare perspective - of different options for waste disposal and energy recovery cannot be drawn before the external damage of these practices are evaluated accordingly.

**Table 1.** Summary of burdens and impact for a MSW incinerator

| Burden | Receptor | Target | Impact | Range of Uncertainty* |
|---|---|---|---|---|
| TSP, Sulfates, Nitrates | air | Public | Health | A-B |
| Cd and pCDD, pCDF | air | Public | Health (cancer) | B |
| Hg, Pb, HCl | air | Public | Health | nq |
| HCl, HF | air | Workers | Health | nq |
| TSP, Cd, Hg, Pb, CDD, CDF | soil | Public | Health | nq |
| Plant construction | - | Workers | Accidents | B |
| $CO_2$ | air | World | Global warming | C |
| $SO_2$ | air | Agriculture | Crop damage | nq |
| $SO_x$, $NO_x$ | air | Buildings | Materials corrosion | nq |
| $SO_x$, $NO_x$, HCl, HF | air | Forests, lakes | Acidification | nq |
| Ozone (via $NO_x$) | air | Public | Health | B |
| Ozone (via $NO_x$) | air | Agriculture | Crop damage | B |
| Visual impact | - | Public | Amenity | B |
| Visual impact | - | Residents | Estate value losses | B |

(*) Uncertainty rating: **A** = high confidence ($\sigma_g$ = 2.5 - 4),
**B** = medium confidence ($\sigma_g$ = 4 - 6),
**C** = low confidence ($\sigma_g$= 6 - 12)

nq = not quantified
$\sigma_g$ = geometric standard deviation

**Table 2.** Summary of burdens and impact pathways for a MSW landfill site

| Burden | Receptor | Target | Impact | Range of Uncertainty* |
|---|---|---|---|---|
| Windblown bottom ashes | air | Public | Health | nq |
| $CO_2$, $CH_4$ | air | World | Global warming | C |
| $H_2S$, S-organics | air | Forests, lakes | Acidification | nq |
| $H_2S$, S-organics | air | Public | Odours | B |
| PAH (benzene) | air | Public | Health (carcinog.) | B |
| Sulfates, Nitrates | air | Public | Health (morbidity) | A |
| Fires | air | Public | Health | nq |
| Visual impact | | Public | Amenity | B |
| Visual impact | | Residents | Estate value losses | B |
| $SO_2$ | air | Agriculture | Crop damage | nq |
| $SO_x$, $NO_x$ | air | Buildings | Materials corrosion | nq |
| $SO_x$, $NO_x$ | air | Forests, lakes | Acidification | nq |
| Ozone (via $NO_x$) | air | Public | Health | B |
| Ozone (via $NO_x$) | air | Agriculture | Crop damage | B |
| MSW transport | air, accidents | Public | Traffic | A-B |
| Pathogenic dissemination by animals | | Public | Health | nq |
| Construction | | Workers | Accidents | nq |

(*) Uncertainty rating: **A** = high confidence (sg = 2.5 - 4),
**B** = medium confidence ($\sigma_g$ = 4 - 6),
**C** = low confidence ($\sigma_g$= 6 - 12)

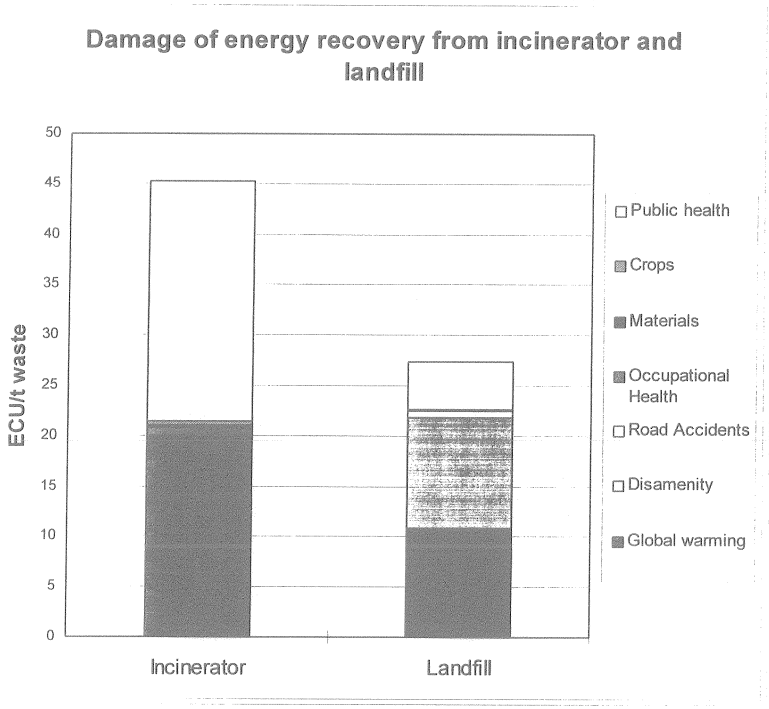



**Figure 1.** Total damage arising from waste incineration and landfilling in terms of ECU per tonne of waste disposed

# EXTERNE

EXTERNALITIES
OF
ENERGY

Vol 9: FUEL CYCLES FOR EMERGING AND END-USE TECHNOLOGIES, TRANSPORT & WASTE

# PART I

# EXTERNAL COSTS OF END-USE, COGENERATION AND NEW TECHNOLOGIES

# EXTERNAL COSTS OF END-USE, COGENERATION AND NEW TECHNOLOGIES

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# EXTERNAL COSTS OF END-USE, COGENERATION AND NEW TECHNOLOGIES

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# 1. INTRODUCTION

## 1.1 Objectives of the project

The use of energy causes damage to a wide range of receptors, including human health, natural ecosystems, and the built environment. Such damages are referred to as external costs, as they are not reflected in the market price of energy. These externalities have been traditionally ignored.

However, there is a growing interest towards the internalisation of externalities to assist policy and decision making. Several European and international organisms have expressed their interest in this issue, as may be seen in the 5th Environmental Action Programme, in the White Paper on Growth, competitiveness and employment, or the White Paper on Energy, all from the European Commission.

This interest has led to the development of internationally agreed tools for the evaluation of externalities, and to its application to different energy sources.

Within the European Commission R&D Programme Joule II, the EXTERNE Project developed and demonstrated a unified methodology for the quantification of the externalities of different power generation technologies. Launched in 1991 as a collaborative project with the US-DOE, and continued afterwards by the EC as the EXTERNE project, it has involved more then 40 different European institutes from 9 countries, as well as scientists from the US. This resulted in the first comprehensive attempt to use a consistent 'bottom-up' methodology to evaluate the external costs associated with a wide range of different fuel cycles. The result was identified by both the European and American experts in this field as currently the most advanced project world-wide for the evaluation of external costs of power generation.

Under Joule III, this project has been continued with three distinguished major tasks: EXTERNE Core for the further development and updating of the methodology, EXTERNE National Implementation to create an EU-wide data set and EXTERNE-Transport for the application of the EXTERNE methodology to energy related impacts from transport. The current report is part of the EXTERNE Core project.

The objective of this sub-task of the EXTERNE Core project is to apply the existing framework to new domains, as there are:

- end-use technologies:
  - domestic space heating with oil, natural gas and electricity, and
  - demand-side options (here with respect to refrigerators, windows and heat exchangers);
- cogeneration; and

- future technologies (here IGCC, fluidised bed combustion, fuel cells for stationary applications and use of geothermal energy).

Thereby, new methodological issues arise in some cases and in others new priority impact pathways have to be analysed, thus extending the existing accounting framework.

In spite of all the uncertainties related to the externalities assessment, the output of the project might prove to be very useful for policy-making, both at the national and EU level. The results obtained provide a good basis to start the study of the internalisation of the external costs of energy, which has been frequently cited as one of the objectives of EU energy policy. Other possibility is to use the results for comparative purposes. The site sensitivity of the externalities might encourage the application of the methodology for the optimisation of site selection processes, or for cost-benefit analysis of the introduction of cleaner technologies.

## 1.2 Structure of this report

The structure of this report reflects that it is part of a wider set of publications. Thus, methodological issues and priority impact pathways are only discussed in more detail in the main body of the report if they are not covered already by the existing framework, which is described in seven appendices. In these appendices the major inputs from different scientific disciplines into the framework (e.g. information on dose-response functions) are summarised. A full-lenght discussion of the EXTERNE methodology can be found in the methodology report of 1995 (European Commission, 1995) and the up-coming methodology report (intended to be published in 1998) (European Commission, 1998).

The EXTERNE methodology in general is outlined in the following chapter. Chapter 3 covers end-use technologies, i.e. domestic-space heating and demand-side options, in chapter 4 issues related to cogeneration are discussed, and in chapter 5 new technologies are analysed.

## 1.3 References

European Commission (1995): EUR 16521 - EXTERNE: Externalities of Energy - Vol. 2: Methodology. Office for Official Publications of the European Commission, Luxembourg

# 2. METHODOLOGY

## 2.1 Approaches Used for Externality Analysis

The EXTERNE Project uses the 'impact pathway' approach for the assessment of the external impacts and associated costs resulting from the supply and use of energy. The analysis proceeds sequentially through the pathway, as shown in Figure 2.1. Emissions and other types of burden such as risk of accident are quantified and followed through to impact assessment and valuation. The approach thus provides a logical and transparent way of quantifying externalities.

However, this style of analysis has only recently become possible, through developments in environmental science and economics, and improvements in computing power has. Early externalities work used a 'top-down' approach (the impact pathway approach being 'bottom-up' in comparison). Such analysis is highly aggregated, being carried out at a regional or national level, using estimates of the total quantities of pollutants emitted or present and estimates of the total damage that they cause. Although the work of Hohmeyer (1988) and others advanced the debate on externalities research considerably, the style of analysis was too simplistic for adoption for policy analysis. In particular, no account could be taken of the dependence of damage with the location of emission, beyond minor corrections for variation of income at the valuation stage.

An alternative approach was the 'control cost' method, which substitutes the cost of reducing emissions of a pollutant (which are determined from engineering data) for the cost of damages due to these emissions. Proponents of this approach argued that when elected representatives decide to adopt a particular level of emissions control they express the collective 'willingness-to-pay' of the society that they represent to avoid the damage. However, the method is entirely self-referencing - if the theory was correct, whatever level of pollution abatement is agreed would by definition equal the economic optimum. Although knowledge of control costs is an important element in formulating prescriptive regulations, presenting them as if they were damage costs is to be avoided.

Life cycle analysis (OECD, 1992; Heijungs *et al*, 1992; Lindfors *et al*, 1995) is a flourishing discipline whose roots go back to the net energy analyses that were popular twenty years ago. While there are several variations, all life cycle analysis is in theory based on a careful and holistic accounting of all energy and material flows associated with a system or process. The approach has typically been used to compare the environmental impacts associated with different products that perform similar functions, such as plastic and glass bottles. Restriction of the assessment to material and energy flows means that some types of externality (such as the fiscal externalities arising from energy security) are completely outside the scope of LCA.

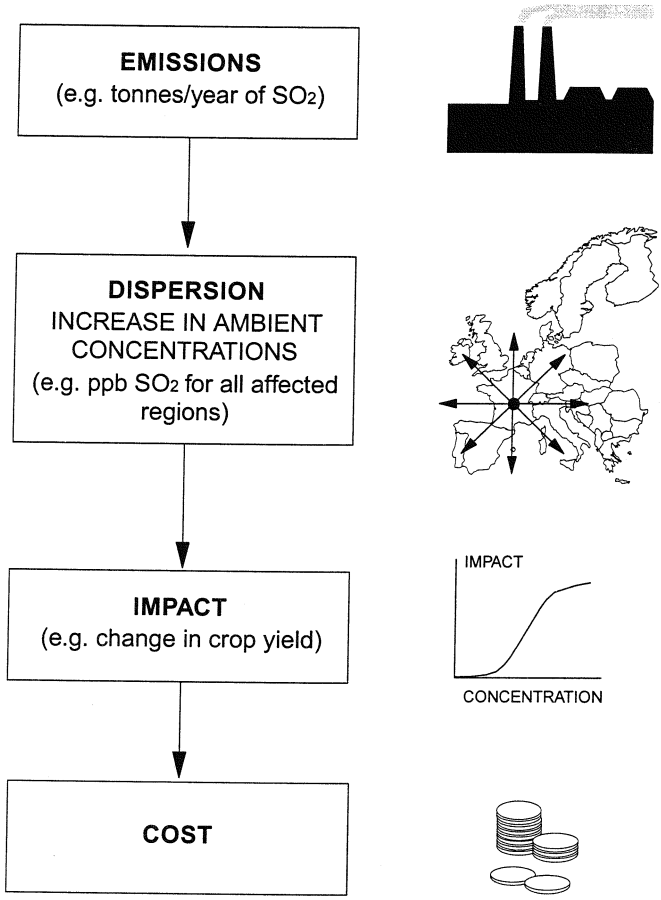



**Figure 2.1** An illustration of the main steps of the impact pathways methodology applied to the consequences of pollutant emissions. Each step is analysed with detailed process models.

The EXTERNE method has numerous links to LCA. The concept of fuel cycle or fuel chain analysis, in which all components of a given system are analysed 'from cradle to grave', corresponds with the LCA framework. Hence for electric power fuel chains the analysis undertaken within the EXTERNE Project covers (so far as possible); fuel extraction, transportation

and preparation of fuels and other inputs; plant construction, plant operation (power generation), waste disposal and plant decommissioning.

There are, however, some significant differences between externalities analysis as presented in this study and typical LCA analysis. Life cycle analyses tend not to be specific on the calculation of impacts, if they have attempted to quantify impacts at all. For example, the 'classification factors' identified by Heijungs *et al* (1992) for each pollutant are independent of the site of release. For air pollution these factors were calculated with the assumption of uniform mixing in the earth's atmosphere. While this can be justified for greenhouse gases and other pollutants with long residence times, it is unrealistic for particulate matter, $NO_x$, $SO_2$ and ozone ($O_3$). The reason for this radical approximation lies in the choice of emphasis in LCA: accounting for all material flows, direct and induced. Since induced flows occur at many geographically different points under a variety of different conditions, it is simply not practicable to model the fate of all emissions. In this sense, EXTERNE is much more ambitious and precise in its estimates than LCA.

A second difference is that most LCA studies have a much more stringent view on system boundaries and do not prioritise between different impacts. The EXTERNE analysts have to a large extent decided themselves if certain stages of the fuel cycle, such as plant construction or fuel transportation, can be excluded. Such decisions are made from experience of the likely magnitude of damages, and a knowledge of whether a given type of impact is *perceived* to be serious. [Note that it is recommended to quantify damages for any impact perceived to be serious whether or not earlier analysis has suggested that associated damages will be negligible]. What might be referred to as analytical 'looseness' is a consequence of the remit of the EXTERNE project, which has as a final objective quantification of the externalities of energy systems. As such the main emphasis of the study is quite properly on the impacts that are likely (given current knowledge) to dominate the results. Externalities assessments based on the EXTERNE methodology but conducted for other purposes may need to take a more truly holistic perspective than has been attempted here.

The analysis presented in this report places its emphasis on the quantification of impacts and cost because people care more about impacts than emissions. The quantification of emissions is merely a step in the analysis. From this perspective the choice between externalities assessment and conventional LCA is a matter of accuracy; uncertainties increase the further the analysis is continued. In general terms, however, it is our view that the fuel chain analyses of the EXTERNE Project can be considered a particular example of life cycle analysis.

## 2.2 Guiding Principles in the Development of the EXTERNE Methodology

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

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

***Consistency***, of methodology, models and assumptions (e.g. system boundaries, exposure-response functions and valuation of risks to life) to allow valid comparisons to be made between different fuel chains and different types of impact within a fuel chain.

***That analysis should be comprehensive***, we should seek to at least identify all of the effects that may give rise to significant externalities, even if some of these cannot be quantified in either physical or monetary terms.

In order to comply with these principles, much of the analysis described in this report looks at the effects of individual power projects which are closely specified with respect to:

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

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

## 2.3 Defining the Boundaries of the Analysis

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

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

### 2.3.1 Stages of the fuel chain

For any project associated with electricity generation the system is centred on the generation plant itself. However, the system boundaries should be drawn so as to account for all potential effects of a fuel chain. The exact list of stages is clearly dependent on the fuel chain in question, but would include activities linked to the manufacture of materials for plant, construction, demolition and site restoration as well as power generation. Other stages may need to be considered, such as, exploration, extraction, processing and transport of fuel, and the generation of wastes and by-products, and their treatment prior to disposal.

In practice, a complete analysis of each stage of a fuel chain is often not necessary in order to meet the objectives of the analysis (see below). However, the onus is on the analyst to demon-

strate that this is the case - it cannot simply be assumed. Worth noting is the fact that variation in laws and other local conditions will lead to major differences between the importance of different stages in different parts of the world.

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

The treatment of waste matter and by-products deserves special mention. Impacts associated with waste sent for disposal are part of the system under analysis. However, impacts associated with waste utilised elsewhere (which are here referred to not a waste but as by-products) should be considered as part of the system to which they are transferred from the moment that they are removed from the boundaries of the fuel chain. It is of course important to be sure that a market exists for any such by-products. The capacity of, for example, the building industry to utilise gypsum from flue gas desulphurisation systems is clearly finite. If it is probable that markets for particular by-products are already saturated, the 'by-product' must be considered as waste instead. A further difficulty lies in the uncertainties about future management of waste storage sites. For example, if solid residues from a power plant are disposed in a well engineered and managed landfill there is no impact (other than land use) as long as the landfill is correctly managed; however, for the more distant future such management is not certain.

### 2.3.2 Location of fuel chain activities

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

- Variation in technology arising from differing legal requirements (e.g. concerning the use of pollution abatement techniques, occupational safety standards, etc.);
- Variation in fuel quality;
- Variations in atmospheric dispersion;
- Differences in the sensitivity of the human and natural environment upon which fuel chain burdens impact.

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

However, the selection of a particular series of sites for a particular fuel chain is not altogether realistic, particularly in relation to upstream impacts. For example, although some coal fired power stations use coal from the local area, an increasing number use coal imported from a number of different countries. This has now been taken into account.

### 2.3.3 Identification of fuel chain technologies

The main objective of this project was to quantify the external costs of power generation technologies built in the 1990s. For the most part it was not concerned with future technologies that are as yet unavailable, nor with older technologies which are gradually being decommissioned.

Over recent years an increasingly prescriptive approach has been taken to the regulation of new power projects. The concept of Best Available Techniques (BAT), coupled with emission limits and environmental quality standards defined by both national and international legislation, restrict the range of alternative plant designs and rates of emission. This has made it relatively easy to select technologies for each fuel chain on a basis that is consistent across fuel chains. However, care is still needed to ensure that a particular set of assumptions are valid for any given country. Across the broader EXTERNE National Implementation Project particular variation has for example been found with respect to the control of $NO_x$ in different EU Member States.

As stated above, the present report deals mainly with closely specified technology options. Results have also been aggregated for the whole electricity generating sector, providing first estimates of damages at the national level.

### 2.3.4 Identification of fuel chain burdens

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

- Solid wastes;
- Liquid wastes;
- Gaseous and particulate air pollutants;
- Risk of accidents;
- Occupational exposure to hazardous substances;
- Noise;
- Others (e.g. exposure to electro-magnetic fields, emissions of heat).

During the identification of burdens no account has been taken of the likelihood of any particular burden actually causing an impact, whether serious or not. For example, in spite of the concern that has been voiced in recent years there is no definitive evidence that exposure to electro-magnetic fields associated with the transmission of electricity is capable of causing harm. The purpose of the exercise is simply to catalogue everything to provide a basis for the analysis of different fuel chains to be conducted in a consistent and transparent manner, and to

provide a firm basis for revision of the analysis as more information on the effects of different burdens becomes available in the future.

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

### 2.3.5 Identification of impacts

The next part of the work involves identification of the potential impacts of these burdens. At this stage it is irrelevant whether a given burden will actually cause an appreciable impact; all potential impacts of the identified burdens should be reported. The emphasis here is on making analysts demonstrate that certain impacts are of little or no concern, according to current knowledge. The conclusion that the externalities associated with a particular burden or impact, when normalised to fuel chain output, are likely to be negligible is an important result that should not be passed over without comment. It will not inevitably follow that action to reduce the burden is unnecessary, as the impacts associated with it may have a serious effect on a small number of people. From a policy perspective it might imply, however, that the use of fiscal instruments might not be appropriate for dealing with the burden efficiently.

The first series of EXTERNE reports (European Commission, 1995a-f) provided comprehensive listings of burdens and impacts for most of the fuel chains considered. The tasks outlined in this section and the previous one are therefore not as onerous as they seem, and will become easier with the development of appropriate databases.

### 2.3.6 Valuation criteria

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

### 2.3.7 Spatial limits of the impact analysis

The system boundary also has spatial and temporal dimensions. Both should be designed to capture impacts as fully as possible.

This has major implications for the analysis of the effects of air pollution in particular. It necessitates extension of the analysis to a distance of hundreds of kilometres for many air pol-

lutants operating at the 'regional' scale, such as ozone, secondary particles, and $SO_2$. For greenhouse gases the appropriate range for the analysis is obviously global. Consideration of these ranges is in marked contrast to the standard procedure employed in environmental impact assessment which considers pollutant transport over a distance of only a few kilometres and is further restricted to primary pollutants. The importance of this issue in externalities analysis is that in many cases in the EXTERNE Project it has been found that regional effects of air pollutants like $SO_2$, $NO_x$ and associated secondary pollutants are far greater than effects on the local scale (for examples see European Commission, 1995c). In some locations, for example close to large cities, this pattern is reversed, and accordingly the framework for assessing air pollution effects developed within the EcoSense model allows specific account to be taken of local range dispersion.

It is frequently necessary to truncate the analysis at some point, because of limits on the availability of data. Under these circumstances it is recommended that an estimate be provided of the extent to which the analysis has been restricted. For example, one could quantify the proportion of emissions of a given pollutant that have been accounted for, and the proportion left unaccounted.

### 2.3.8 Temporal limits of the impact analysis

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

The approach adopted here is that discounting should only be applied after costs are quantified. The application of any discount rate above zero can reduce the cost of major events in the distant future to a negligible figure. This perhaps brings into question the logic of a simplistic approach to discounting over time scales running far beyond the experience of recorded history. There is clear conflict here between some of the concepts that underlie traditional economic analysis and ideas on sustainability over timescales that are meaningful in the context of the history of the planet. For further information, the discounting of global warming damages is discussed further in Appendix V.

The assessment of future costs is of course not simply a discounting issue. A scenario based approach is also necessary in some cases in order to describe the possible range of outcomes. This is illustrated by the following examples;

- A richer world would be better placed to take action against the impacts of global warming than a poorer one;
- The damages attributable to the nuclear fuel chain could be greatly reduced if more effective treatments for cancer are discovered.

Despite the uncertainties involved it is informative to conduct analysis of impacts that take effect over periods of many years. By doing so it is at least possible to gain some idea of how

important these effects might be in comparison to effects experienced over shorter time scales. The chief methodological and ethical issues that need to be addressed can also be identified. To ignore them would suggest that they are unlikely to be of any importance.

## 2.4 Analysis of Impact Pathways

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

- Prioritisation of impacts;
- Description of priority impact pathways;
- Quantification of burdens;
- Description of the receiving environment;
- Quantification of impacts;
- Economic valuation;
- Description of uncertainties.

### 2.4.1 Prioritisation of impacts

It is possible to produce a list of several hundred burdens and impacts for many fuel chains (see European Commission, 1995c, pp. 49-58). A comprehensive analysis of all of these is clearly beyond the scope of externality analysis. In the context of this study, it is important to be sure that the analysis covers those effects that (according to present knowledge) will provide the greatest externalities (see the discussion on life cycle analysis in section 2.1). Accordingly, the analysis presented here is limited, though only after due consideration of the potential magnitude of all impacts that were identified for the fuel chains that were assessed. It is necessary to ask whether the decision to assess only a selection of impacts in detail reduces the value of the project as a whole. We believe that it does not, as it can be shown that many impacts (particularly those operating locally around any given fuel chain activity) will be negligible compared to the overall damages associated with the technology under examination.

There are good reasons for believing that local impacts will tend to be of less importance than regional and global effects. The first is that they tend to affect only a small number of people. Even though it is possible that some individuals may suffer very significant damages these will not amount to a significant effect when normalised against a fuel chain output in the order of several Tera-Watt ($10^{12}$ Watt) hours per year. It is likely that the most appropriate means of controlling such effects is through local planning systems, which be better able than policy developed using externalities analysis to deal flexibly with the wide range of concerns that may exist locally. A second reason for believing that local impacts will tend to be less significant is that it is typically easier to ascribe cause and effect for impacts effective over a short range than for those that operate at longer ranges. Accordingly there is a longer history of legislation to combat local effects. It is only in recent years that the international dimension of pollution of the atmosphere and water systems has been realised, and action has started to be taken to deal with them.

There are obvious exceptions to the assertion that in many cases local impacts are of less importance than others;

- Within OECD states one of the most important exceptions concerns occupational disease, and accidents that affect workers and members of the public. Given the high value attached to human life and well-being there is clear potential for associated externalities to be large.
- Other cases mainly concern renewable technologies, at least in countries in which there is a substantial body of environmental legislation governing the design and siting of nuclear and fossil-fired plant. For example, most concern over the development of wind farms typically relates to visual intrusion in natural landscapes and to noise emissions.
- There is the possibility that a set of conditions - meteorology, geography, plant design, proximity of major centres of population, etc. - can combine to create local air quality problems.

The analysis of certain upstream impacts appears to create difficulties for the consistency of the analysis. For example, if we treat emissions of $SO_2$ from a power station as a priority burden, why not include emissions of $SO_2$ from other parts of the fuel chain, for example from the production of the steel and concrete required for the construction of the power plant? Calculations made in the early stages of EXTERNE using databases, such as GEMIS (Fritsche *et al*, 1992), showed that the emissions associated with material inputs to fossil power plants are 2 or 3 orders of magnitude lower than those from the power generation stage. It is thus logical to expect that the impacts of such emissions are trivial in comparison, and can safely be excluded from the analysis - if they were to be included the quantified effects would be secondary to the uncertainties of the analysis of the main source of emissions. However, this does not hold across all fuel chains. In the reports on both the wind fuel chain (European Commission, 1995f) and the photovoltaic fuel chain (ISET, 1995), for example, it was found that emissions associated with the manufacture of plant are capable of causing significant externalities, relative to the others that were quantified.

The selection of priorities partly depends on whether one wants to evaluate damages or externalities. In quite a few cases the externalities are small in spite of significant damages. For example, if a power plant has been in place for a long time, much of the externality associated with visual and noise impacts will have been internalised through adjustments in the price of housing. It has been argued that occupational health effects are also likely to be internalised.

For example, if coal miners are rational and well informed their work contracts should offer benefits that internalise the incremental risk that they are exposed to. However, this is a very controversial assumption, as it depends precisely upon people being both rational and well informed and also upon the existence of perfect mobility in labour markets. For the present time we have quantified occupational health effects in full, leaving the assessment of the degree to which they are internalised to a later date.

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

### 2.4.2 Description of priority impact pathways

Some impact pathways analysed in the present study are extremely simple in form. For example, the construction of a wind farm will affect the appearance of a landscape, leading to a change in visual amenity. In other cases the link between 'burden' (defined here simply as something that causes an 'impact') and monetary cost is far more complex. To clearly define the linkages involved in such cases we have drawn a series of diagrams. One of these is shown in Figure 2.2, illustrating the series of processes that need to be accounted for from emission of acidifying pollutants to valuation of impacts on agricultural crops. It is clearly far more complex than the pathway suggested by Figure 2.1.

A number of points should be made about Figure 2.2. It (and others like it) do not show what has been carried out within the project. Instead they illustrate an ideal - what one would like to do if there was no constraint on data availability. They can thus be used both in the development of the methodology and also as a check once analysis has been completed, to gain an impression of the extent to which the full externality has been quantified. This last point is important because much of the analysis presented in this report is incomplete. This reflects on the current state of knowledge of the impacts addressed. The analysis can easily be extended once further data becomes available. Also, for legibility, numerous feedbacks and interactions are not explicitly shown in the diagrammatic representation of the pathway.

### 2.4.3 Quantification of burdens

The data used to quantify burdens must be both *current* and *relevant* to the situation under analysis. Emission standards, regulation of safety in the workplace and other factors vary significantly over time and between and within different countries. It is true that the need to meet these demands creates difficulties for data collection. However, given that the objective of this work is to provide as far as possible an accurate account of the environmental and social burdens imposed by energy supply and use, these issues should not be ignored. It is notable that data for new technologies can change rapidly following their introduction. In addition to the inevitable refinement of technologies over time, manufacturers of novel equipment may be cautious in their assessment of plant performance. As an example of this latter point, $NO_x$ emission factors for combined cycle gas turbine plant currently coming on stream in several countries are far lower than was suggested by Environmental Statements written for the same plant less than five years ago.

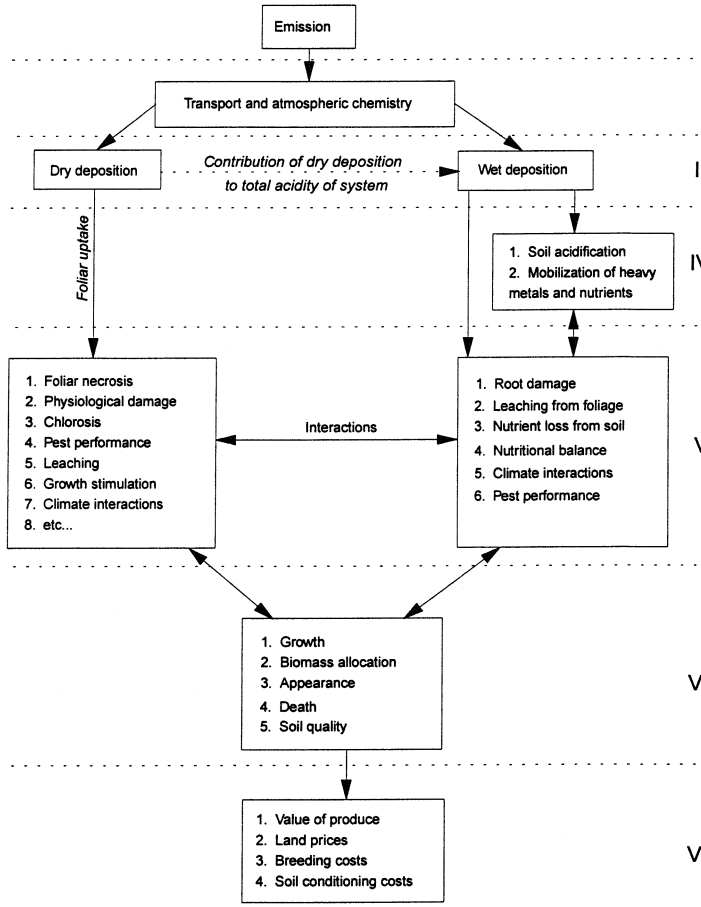



**Figure 2.2** The impact pathway showing the series of linkages between emission of acidifying pollutants and ozone precursors and valuation of impacts on agricultural systems.

All impacts associated with pollution of some kind require the quantification of emissions. Emission rates of the 'classical' air pollutants ($CO_2$, $SO_2$, $NO_x$, CO, volatile organic compounds and particulate matter) are quite well known. Especially well determined is the rate of $CO_2$ emission for fuel using equipment; it depends only on the efficiency of the equipment and the carbon/hydrogen ratio of the fuel - uncertainty is negligible. Emissions of the other

classical air pollutants are somewhat less certain, particularly as they can vary with operating conditions, and maintenance routines. The sulphur content of different grades of oil and coal can vary by an order of magnitude, and hence, likewise, will emissions unless this is compensated for through varying the performance of abatement technologies. The general assumption made in this study is that unless otherwise specified, the technology used is the best available according to the regulations in the country of implementation, and that performance will not degrade. We have sought to limit the uncertainty associated with emissions of these pollutants by close identification of the source and quality of fuel inputs within the study.

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

The problem becomes more difficult for the upstream and downstream stages of the fuel chain because of the variety of technologies that may be involved. Particularly important may be some stages of fuel chains such as biomass, where the fuel chain is potentially so diverse that it is possible that certain activities are escaping stringent environmental regulation.

The burdens discussed so far relate only to routine emissions. Burdens resulting from accidents also need to be considered. These might result in emissions (e.g. of oil) or an incremental increase in the risk of injury or death to workers or members of the public. Either way it is normally necessary to rely upon historical data to quantify accident rates. Clearly the data should be as recent as possible so that the rates used reflect current risks. Major uncertainty however is bound to be present when extreme events need to be considered, such as the disasters at Chernobyl and on the Piper Alpha oil rig in the North Sea. To some extent it is to be expected that accident rates will fall over time, drawing on experience gained. However, structural changes in industries, for example through privatisation or a decrease in union representation, may reverse such a trend.

Wherever possible data should be relevant to the country where a particular fuel chain activity takes place. Major differences in burdens may arise due to different standards covering occupational health, extension of the distance over which fuel needs to be transported, etc.

### 2.4.4 Description of the receiving environment

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

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

The range of the reference environment for any impact requires expert assessment of the area influenced by the burden under investigation. As stated above, arbitrary truncation of the reference environment is methodologically wrong and will produce results that are incorrect. It is to be avoided as far as possible.

Clearly the need to describe the sensitivity of the receiving environment over a vast area (extending to the whole planet for some impacts) creates a major demand on the analyst. This is simplified by the large scale of the present study - which has been able to draw on data held in many different countries. Further to this it has been possible to draw on numerous databases that are being compiled as part of other work, for example on critical loads mapping. Databases covering the whole of Europe, describing the distribution of the key receptors affected by $SO_2$, $NO_x$, $NH_3$ and fine particles have been derived or obtained for use in the EcoSense software developed by the study team.

In order to take account of future damages, some assumption is required on the evolution of the stock at risk. In a few cases it is reasonable to assume that conditions will remain roughly constant, and that direct extrapolation from the present day is as good an approximation as any. In other cases, involving for example the emission of acidifying gases or the atmospheric concentration of greenhouse gases this assumption is untenable, and scenarios need to be developed. Confidence in these scenarios clearly declines as they extend further into the future.

### 2.4.5 Quantification of impacts

The methods used to quantify various types of impact are discussed in depth in the report on the study methodology (European Commission, 1998). The functions and other data that we have used are summarised at the back of this report in Appendices I (describing the EcoSense software), II (health), III (materials), IV (ecological receptors), V (global warming effects) and VI (other impacts), VII (economic issues) and VIII (uncertainty). The complexity of the analysis varies greatly between impacts. In some cases externalities can be calculated by multiplying together as few as 3 or 4 parameters. In others it is necessary to use a series of sophisticated models linked to large databases.

Common to all of the analysis conducted on the impacts of pollutants emitted from fuel chains is the need for modelling the dispersion of pollutants and the use of a dose-response function of some kind. Again, there is much variation in the complexity of the models used (see Appendix I). The most important pollutant transport models used within EXTERNE relate to the atmospheric dispersion of pollutants. They need to account not only for the physical transport of pollutants by the winds but also for chemical transformation. The dispersion of pollutants that are in effect chemically stable in the region of the emission can be predicted using Gaussian plume models. These models assume source emissions are carried in a straight line by the wind, mixing with the surrounding air both horizontally and vertically to produce pollutant

concentrations with a normal (or Gaussian) spatial distribution. The use of these models is typically constrained to within a distance of 100 km of the source.

Air-borne pollutant transport of course extends over much greater distances than 100 km. A different approach is needed for assessing regional transport as chemical reactions in the atmosphere become increasingly important. This is particularly so for the acidifying pollutants. For this analysis we have used receptor-orientated Lagrangian trajectory models. The outputs from the trajectory models include atmospheric concentrations and deposition of both the emitted species and secondary pollutants formed in the atmosphere.

A major problem has so far been the lack of a regional model of ozone formation and transport within fossil-fuel power station plumes that is applicable to the European situation. In consequence a simplified approach has been adopted for assessment of ozone effects (European Commission, 1998).

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

Dose-response functions come in a variety of functional forms, some of which are illustrated in Figure 2.3. They may be linear or non-linear and contain thresholds (e.g. critical loads) or not. Those describing effects of various air pollutants on agriculture have proved to be particularly complex, incorporating both positive and negative effects, because of the potential for certain pollutants, e.g. those containing sulphur and nitrogen, to act as fertilisers.

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

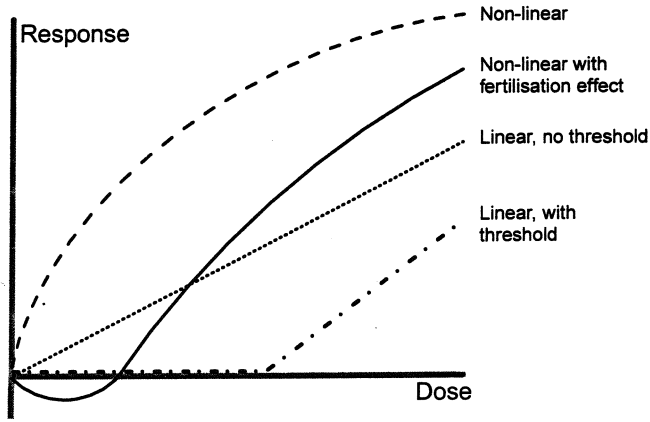



**Figure 2.3** A variety of possible forms for dose-response functions.

The description and implementation of exposure-response relationships is fundamental to the entire EXTERNE Project. Much of the report on methodology (European Commission, 1998) is, accordingly, devoted to assessment of the availability and reliability of these functions.

### 2.4.6 Economic valuation

The rationale and procedures underlying the economic valuation applied within the EXTERNE Project are discussed in Appendix VII and in more detail in the methodology report (European Commission, 1998). The approach followed is based on the quantification of individual 'willingness to pay' (WTP) for environmental benefit.

A limited number of goods of interest to this study - crops, timber, building materials, etc. - are directly marketed, and for these valuation data are easy to obtain. However, many of the more important goods of concern are not directly marketed, including human health, ecological systems and non-timber benefits of forests. Alternative techniques have been developed for valuation of such goods, the main ones being hedonic pricing, travel cost methods and contingent valuation (Appendix VII). All of these techniques involve uncertainties, though they have been considerably refined over the years.

The base year for the valuation described in this report is 1995, and all values are referenced to that year. The unit of currency used is the ECU. The exchange rate was approximately 1 ECU to US$1.25 in 1995.

The central discount rate used for the study is 3%, with upper and lower rates of 0% and 10% also used to show sensitivity to discount rate. The rationale for the selection of this range and best estimate, and a broader description of issues relating to discounting, was given in an earlier report (European Commission, 1995b).

### 2.4.7 Assessment of uncertainty

Uncertainty in externality estimates arises in several ways, including:

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

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

Traditional statistical techniques would ideally be used to describe the uncertainties associated with each of our estimates, to enable us to report a median estimate of damage with an associated probability distribution. Unfortunately this is rarely possible without excluding some significant aspect of error, or without making some bold assumption about the shape of the probability distribution. Alternative methods are therefore required, such as sensitivity analysis, expert judgement and decision analysis. In this phase of the study a more clearly quantified description of uncertainty has been attempted than previously. Further discussion is provided in Appendix VIII, though it is worth mentioning that in this area of work uncertainties tend to be so large that additive confidence intervals usually do not make sense; instead one should specify multiplicative confidence intervals. The uncertainties of each stage of an impact pathway need to be assessed and associated errors quantified. The individual deviations for each stage are then combined to give an overall indication of confidence limits for the impact under investigation.

## 2.5 Priority Impacts Assessed in the EXTERNE Project

### 2.5.1 Fossil technologies

The following list of priority impacts was derived for the fossil fuel chains considered in the earlier phases of EXTERNE. It is necessary to repeat that this list is compiled for the specific fuel chains considered by the present study, and should be reassessed for any new cases. The first group of impacts are common to all fossil fuel chains:

1. Effects of atmospheric pollution on human health;
2. Accidents affecting workers and/or the public;
3. Effects of atmospheric pollution on materials;
4. Effects of atmospheric pollution on crops;
5. Effects of atmospheric pollution on forests;
6. Effects of atmospheric pollution on freshwater fisheries;
7. Effects of atmospheric pollution on unmanaged ecosystems;
8. Impacts of global warming;
9. Impacts of noise.

To these can be added a number of impacts that are fuel chain dependent:

10. Impacts of coal and lignite mining on ground and surface waters;
11. Impacts of coal mining on building and construction;
12. Resettlement necessary through lignite extraction;
13. Effects of accidental oil spills on marine life;
14. Effects of routine emissions from exploration, development and extraction from oil and gas wells.

### 2.5.2 Nuclear technologies

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

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

Impacts on the environment of increased levels of natural background radiation due to the routine releases of radionuclides have not been considered as a priority impact pathway, except partially in the analysis of major accidental releases.

### 2.5.3 Renewable technologies

The priority impacts for renewables vary considerably from case to case. Each case is dependent upon the local conditions around the implementation of each fuel chain. For the wind fuel chain (European Commission, 1995f) the following were considered:

1. Accidents affecting the public and/or workers;
2. Effects on visual amenity;
3. Effects of noise emissions on amenity;
4. Effects of atmospheric emissions related to the manufacture of turbines and construction and servicing of the site.

Whilst for the hydro fuel chain (European Commission, 1995f) another group was considered:

1. Occupational health effects;
2. Employment benefits and local economic effects;
3. Impacts of transmission lines on bird populations;
4. Damages to private goods (forestry, agriculture, water supply, ferry traffic);
5. Damages to environmental goods and cultural objects.

### 2.5.4 Related issues

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

## 2.6 Summary

This Chapter has introduced the 'impact pathway' methodology of the EXTERNE Project. The authors believe that it provides the most appropriate way of quantifying externalities because it enables the use of the latest scientific and economic data.

Critical to the analysis is the definition of fuel chain boundaries, relating not only to the different stages considered for each fuel chain, but also to the:

- Location of each stage;
- Technologies selected for each stage;
- Identified burdens;
- Identified impacts;
- Valuation criteria;
- Spatial and temporal limits of impacts.

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

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

The analysis requires the use of numerous models and databases, allowing a logical path to be followed through the impact pathways. The functions and other data originally used by EXTERNE were described in an earlier report (European Commission, 1995b). In the present phase of the study this information has been reassessed and many aspects of it have been updated (see European Commission, 1998). It is to be anticipated that further methodological changes will be needed in the future, as further information becomes available particularly regarding the health effects of air pollution and global warming impacts, which together provide some of the most serious impacts quantified under the study.

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# 3. ANALYSIS OF END-USE TECHNOLOGIES

## 3.1 Domestic Space Heating

### 3.1.1 Introduction

Domestic space heating is an important contributor to final energy use in most countries. For example in 1992 in Germany 20.4% of the final energy use was due to domestic space heating (Eckerle *et al.*, 1995). Thus, it can safely be presumed that domestic space heating contributes considerably to the total external costs of the energy sector. It was the objective of this sub-task to extend the methodology for the assessment of external costs developed so far in the EXTERNE project to the analysis of external costs of domestic space heating.

Figure 3.1 shows the distribution of heating systems across the flats in Germany. Oil and gas central heating are the most often employed options, followed by district heating, coal-fired individual heating units and electric thermal storage heating. However, it is expected that the share of coal-fired heating (central and individual) decreases to about 5% till 2000, oil central heating, district heating and electric thermal storage heating remain on the same level while gas central heating increases to about 38 % (Eckerle *et al.*, 1995). In the following central space heating with oil and gas and electric thermal storage heating are analysed. The focus of the analysis is on modern heating systems with the present state of technology. An example of district heating is analysed in section 4.5.

The objective of domestic space heating is to keep a room on a certain temperature level above the ambient temperature. The produced heat, i.e. the useful energy, needed for a certain heating task depends on many factors *inter alia* consumer behaviour. The ratio of the useful energy to the final energy fed to the heating system, on the other hand, depends on the technical characteristics of the heating installation and in some degree on its management.

There are some analyses how *final energy* demand depends on different factors (FDE, FfE, 1993), (Wagner *et al.*, 1989), (Hilberg, 1993), (Eckerle *et al.*, 1991), (Geiger, 1982). In general the following factors are listed:

- climate and weather (often characterised as degree days),
- the size, shape and function of the building or room to be heated (e.g. floor area, volume to surface area ratio, living room or bedroom),
- the thermic quality of the building (thermal insulation, passive use of solar energy, etc.),
- the kind of utilisation (space heating alone or space and water heating combined),
- the technical characteristics of the heating systems (e.g. fuel type, central or decentral heating),

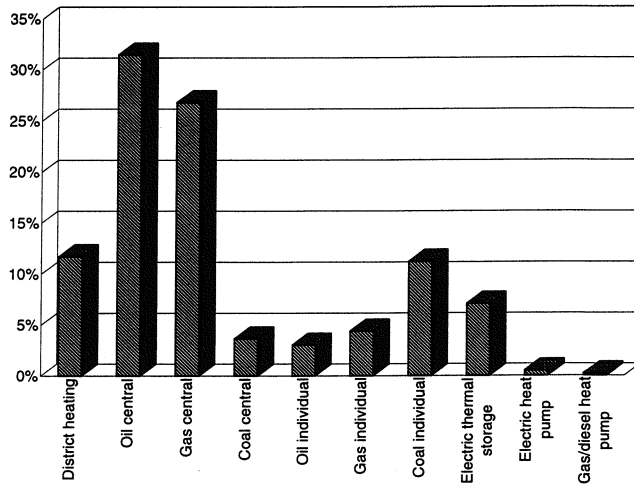



**Figure 3.1** Distribution of heating systems on flats in Germany in 1992 (Eckerle *et al.*, 1995)

- the behaviour of the consumers (management of the heating installation, length of heating period, desired temperature level, heating times and airing habits, etc.), and
- costs (installation and maintenance, fuel, thermal insulation).

However, the analyses do usually not distinguish whether these factors influence
- the demand for certain heating tasks, (rising heating costs may lead e.g. to a decrease in the desired temperature level, at least in some rooms (see e.g. (Hilberg, 1993)),
- the useful energy needed to provide a certain heating task (e.g. the thermic quality of the heated building, building size), and
- the final energy needed to produce the useful energy (e.g. kind of utilisation, technical characteristics, installation costs).

The distinction is often not straightforward, as some factors influence more than one of these steps between final energy supplied and heating task. The weather for example influences the demand for a heating task *and* the useful energy needed. Furthermore, the behaviour of the consumers is again affected by socio-economic factors (e.g. owners or tenants, how heating costs are charged, household type) (Mettler-Meiborn, 1982).

In the past in studies comparing energy use and emissions of different domestic space heating options the quantity 'final energy' was most often used as functional unit (see e.g. (Jansen, 1988), (FDE, FfE, 1993)). The reason is obvious. The final energy, e.g. the oil or gas supplied to the different households, is statistically recorded, the produced heat is not. However, such an approach neglects that different heating options can provide different heating tasks with the same amount of final energy. It is more appropriate to define a certain heating task and to take

the quantity 'useful energy produced' as functional unit. Thus, the quantified external costs depend only on 'technical' factors while factors related to the weather or to consumer behaviour are not relevant. Such an approach has been employed e.g. in the IKARUS project (see e.g. (Gülec *et al.*, 1994), the GEMIS model (Fritsche *et al.*, 1994) and by Drake for the assessment of greenhouse gas emissions (Drake, 1996).

### 3.1.2 Description of Reference Technologies

***Definition of the heating task***

As heating task is selected the central space heating of an one-family house with a flow temperature of 70 °C and full load hours of 1650 h per year. It is assumed that this house is located in Lauffen, 35 km north of Stuttgart.

***Reference Technology for Oil- and Gas-Fired Boilers***

The specified heating task can be met by boilers with a nominal capacity of 20 kW. The technology data are taken from the IKARUS project (Pfitzner, Schäfer, 1994). In that study the ambient temperatures are faded out and the influence of a possible lowering of the temperature during the night is neglected.

As typical oil-fired heating system of the present state of technology a low-temperature boiler with an atomising oil burner (blue-burner) is chosen. The atomising burner breaks the oil up into very small (20–100 µm) droplets before combustion. The atomised fuel is mixed with air and the mixture ignited by an extra-high voltage spark. Blue-burners are a new development that allows a soot-free combustion and reduces $NO_x$ and CO emissions significantly. Contrary to conventional atomising burners in blue burners a part of the flue gases is recirculated to the diffuser. There the flue gas heats the atomised fuel so much that it vaporises and is completely combusted. The burner is named after the resulting blue color of the flame (see Figure 3.2).

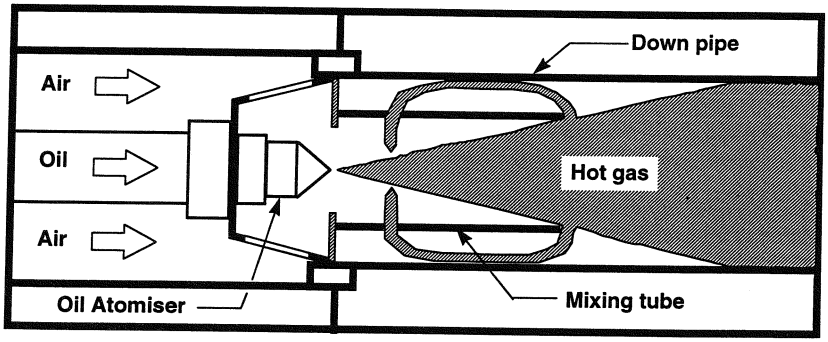
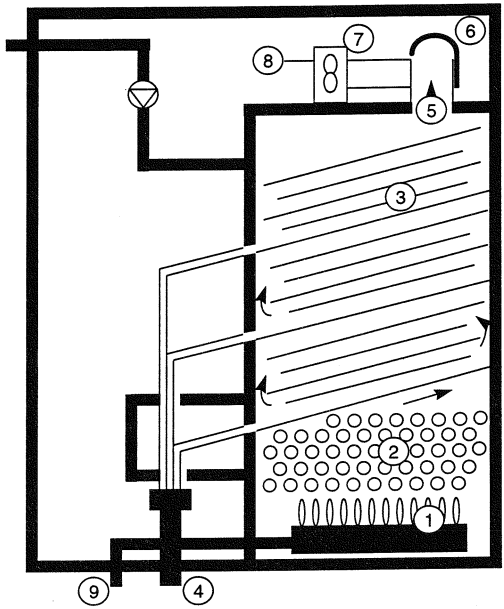



**Figure 3.2** The operation of a blue burner (FIK, 1991b)




1. Atmospheric gas burner
2. Heat exchanger I
3. Heat exchanger II
4. Condensate trap
5. Flue gas outlet
6. Combustion air inlet
7. Combustion air ventilator
8. Combustion air channel
9. Gas input

**Figure 3.3** The set-up of a gas condensing boiler (FIK, 1991a)

For gas a condensing boiler is assumed as at the moment it is the best available technology for gas heating systems with regard to efficiency and emissions. The speciality in the construction of this boiler is that the flue gases preheat the cooled-down water of the return flow, thus reducing the flue gas losses (see Figure 3.3). As the heat of the water vapour is thus recovered efficiencies above 100% are possible (the efficiency is related to the net calorific value of the gas in which the heat of the produced water vapour is not contained).

In Table 3.1 the technical data and the data relevant for the air transport modelling are summarised for both reference heating systems. During the heat distribution energy losses are incurred. Here, a uniform distribution efficiency of 97% is assumed for the fossil-fired boilers (Geiger *et al.*, 1990). In principle, the annual utilisation ratio of the heat generation is lower than the boiler efficiency as during the stand-by availability time losses are incurred. This

**Table 3.1** Technical data of the oil- and gas-fired reference heating systems

| Heating system | Boiler with atomising oil burner (blue-burner) | Gas-fired condensing boiler |
|---|---|---|
| Nominal capacity | 20 kW | 20 kW |
| Boiler efficiency | 93.0% | 95.5% |
| Annual utilisation ratio (incl. distribution losses) | 88.1% | 100.9% |
| Full load hours per year | 1650 h | 1650 h |
| Annual useful energy production | 118.8 GJ | 118.8 GJ |
| *Emissions relative to useful energy and to flue gas volume stream* | | |
| $SO_2$ | 81.7 kg/TJ<br>260.5 mg/Nm$^3$ | 0.5 kg/TJ<br>1.4 mg/Nm$^3$ |
| $NO_x$ | 32.1 kg/TJ<br>102.4 mg/Nm$^3$ | 9.6 kg/TJ<br>33.2 mg/Nm$^3$ |
| Particulates (incl. soot) | 0.6 kg/TJ<br>1.8 mg/Nm$^3$ | 0 kg/TJ |
| CO2 | 84.0 t/TJ | 54.7 t/TJ |
| CO | 5.7 kg/TJ | 5.0 kg/TJ |
| CH4 | 0.7 kg/TJ | 1.1 kg/TJ |
| NMHC | 2.5 kg/TJ | 0.3 kg/TJ |
| $N_2O$ | 1.1 kg/TJ | 0.9 kg/TJ |
| *Data relevant for air transport modelling* | | |
| Stack height | 10 m | 10 m |
| Stack diameter | 0,5 m | 0,5 m |
| Flue gas volume stream (full load) | 21.56 Nm$^3$/h | 20.83 Nm$^3$/h |
| Flue gas temperature | 100 °C | 100 °C |

Sources: Pischinger *et al.* (1994), Pfitzner (1994), FIK (1991a), FIK, (1991b), Fritsche *et al.* (1994), Eckerle *et al.* (1995), Baum (1988)

has changed with the low-temperature boiler with floating boiling water temperatures. With these boilers the annual utilisation ratio can be higher than the boiler efficiency as the waste gas and radiation heat losses are lower at lower boiler water temperatures (Drake, 1996).

***The Upstream Processes of the Fossil-Fired Boilers***

A heating system consists of three components: the final energy supply, the heat generation and the heat distribution. The first comprises all process steps for the production and supply of the final energy needed (here oil or gas). Auxiliary energies for pumps, control, etc. have to be taken into account, too. The upstream processes of an oil power plant and an gas power plant have been analysed in detail based on the model GEMIS 2.1 (Fritsche *et al.*, 1994) employing the database VDEW1.0 compiled by Fichtner Development Engineering (FDE) for VDEW (FDE, 1996). The only major difference between the upstream processes of an oil power plant

and an oil-fired boiler is the distribution of the oil from the refinery to the final user. For the power plant it is done by train, but households are supplied by trucks.

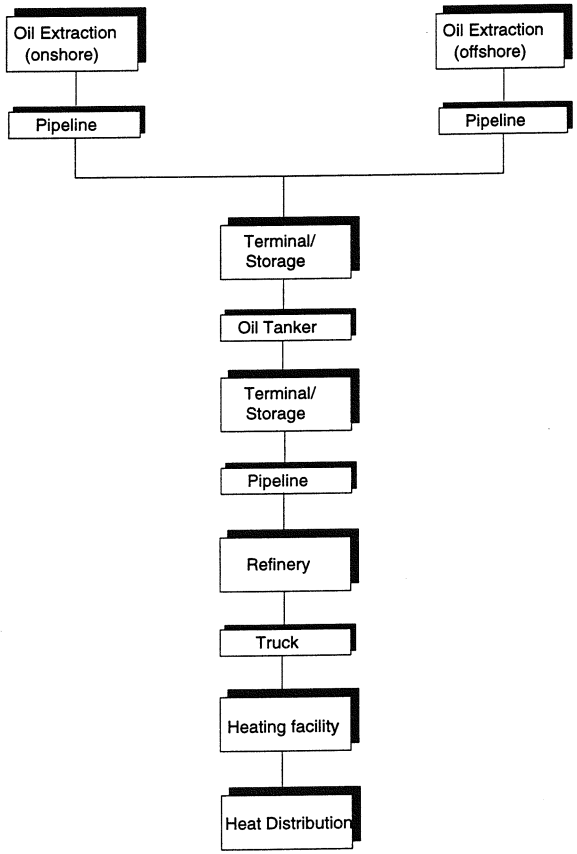



**Figure 3.4** Process steps of the oil-to-domestic space heating fuel cycle

**Table 3.2** Emissions relative to useful energy from the oil-fired boiler and its upstream process steps

| | | Extraction | Transport | Refinery | Boiler | Total |
|---|---|---|---|---|---|---|
| $SO_2$ | [kg/TJ] | 22.0 | 9.3 | $0.4^1+3.9^2$ | 81.7 | 117.3 |
| $NO_x$ | [kg/TJ] | 7.3 | 15.8 | 2.6 | 32.1 | 57.8 |
| TSP | [kg/TJ] | 1.3 | 3.9 | 0.1 | 0.6 | 6.0 |
| $CO_2$ | [t/TJ] | $1.0^1+3.0^2$ | 1.4 | 2.8 | 84.0 | 92.1 |
| CO | [kg/TJ] | 2.8 | 4.5 | 1.2 | 5.7 | 14.2 |
| $CH_4$ | [kg/TJ] | $8.8^1+0.1^2$ | 0.9 | $0.3^1+1.1^2$ | 0.7 | 12.0 |
| NMHC | [kg/TJ] | $5.6^1+0.1^2$ | 0.2 | $0.4^1+0.1^2$ | 2.5 | 8.9 |
| $N_2O$ | [kg/TJ] | 0.07 | 2.4 | 0.05 | 1.1 | 3.6 |

1 Direct emissions

2 Emissions due to energy demand

**Table 3.3** Emissions from the gas-fired boiler and its upstream process steps (maximum leakage in extraction and conditioning in the GUS assumed)

| | | Extraction | Conditioning | Transports | Condensing Boiler | Total |
|---|---|---|---|---|---|---|
| $SO_2$ | [kg/TJ] | $3.3^1 + 0.05^2$ | $14.6^1 + 0.5^2$ | 0.008 | 0.5 | 18.9 |
| $NO_x$ | [kg/TJ] | $1.9^1 + 3.1^2$ | $0.1^1 + 1.7^2$ | 6.3 | 9.6 | 22.6 |
| TSP | [kg/TJ] | 0.04 | $0.008^1 + 2.7^2$ | 0.08 | 0 | 2.9 |
| $CO_2$ | [t/TJ] | $0.3^1 + 0.6^2$ | 0.6 | 1.1 | 54.7 | 57.4 |
| CO | [kg/TJ] | 1.8 | $0.01^1 + 0.8^2$ | 2.9 | 5.0 | 10.6 |
| $CH_4$ | [kg/TJ] | $134.6^1 + 0.2^2$ | $71.9^1 + 0.9^2$ | $179.5^1 + 0.1^2$ | 1.1 | 388.3[3] |
| NMHC | [kg/TJ] | $9.6^1 + 0.1^2$ | $0.1^1 + 0.1^2$ | $0.1^1 + 0.4^2$ | 0.3 | 10.6[4] |
| $N_2O$ | [kg/TJ] | 0.02 | 0.02 | 0.05 | 0.9 | 1.0 |

1 Direct emissions

2 Emissions due to energy demand

3 Total of direct emissions assuming maximum leakage is 386.0 kg $CH_4$ per TJ; assuming minimum leakage total of direct emissions is 233.3 kg/TJ, the total of direct and indirect emissions is 235.6 kg/TJ, then.

4 Total of direct emissions assuming maximum leakage is 9.7 kg NMHC per TJ; assuming minimum leakage total of direct emissions is 1.9 kg/TJ, the total of direct and indirect emissions is 2.8 kg/TJ, then.

FDE based its analysis on the following prognosis of the German oil supply between 1995 and 2000: most of the oil used in Germany comes from the OPEC countries (42%), other important supply countries are Norway and Great Britain (together 34%) and the GUS (22%), only 2% are extracted in Germany itself.

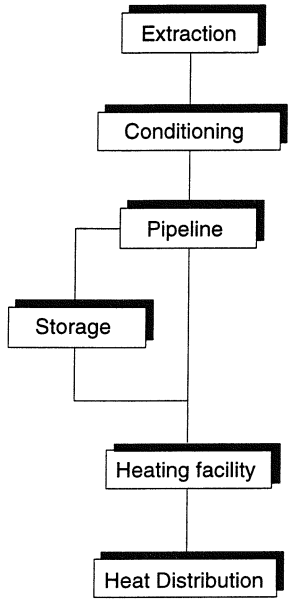



**Figure 3.5** Process steps of the gas-to-domestic space heating fuel cycle

The crude oil is transported by combinations of tanker and pipeline from the extraction countries via the Netherlands, France and Italy to the refinery in Germany. After the refinery process the oil is transported by trucks to the households. Figure 3.4 shows the process steps of the oil fuel cycle for domestic space heating. Table 3.2 and Figure 3.6 summarise the emissions from the oil-to-domestic space heating fuel cycle.

For gas the analysis is based on the following prognosis of the German gas supply between 1995 and 2000: 32% from the GUS, 28% from Norway, 22% from the Netherlands, 2.5% from Denmark, 0.5% from the UK, 18% are extracted in Germany itself. The VDEW 1.0 database treats the Danish and the British in the assessment like the Norwegian gas. After the extraction the gas is conditioned (desulfurisation, drying) and then transported by pipelines to Germany. In Germany it is distributed via distribution networks to the final users. Storages are needed to level daily fluctuations and seasonal differences. While power plants are directly connected to the high pressure network, the households are supplied by the low pressure network connected to the high pressure network. Figure 3.5 shows the process steps of the gas fuel cycle for domestic space heating.

The leakages from gas extraction, conditioning and pipeline transport in the GUS are highly contented. Official data does not exist, several authors, however, have estimated the order of

magnitude of the leakage in the last years. The results differ widely, from 0,25 to 4,5% for gas extraction and conditioning and from 0,07 to 6% for the pipeline transport (Frischknecht *et al.*, 1996). The leakage rates assumed in the VDEW 1.0 database for gas extraction and conditioning in the GUS are at the upper end of the estimates presented so far. Hence, results have also calculated for the leakage rates given in 'Oekoinventare' (Frischknecht *et al.*, 1996), which are at the lower end. As there are some discrepancies in the VDEW database, data on direct emissions due to the leakage from (Frischknecht *et al.*, 1996) are used instead of the data given in VDEW 1.0. Table 3.3 and Figure 3.8 summarise the emissions from the different process steps of the gas-to-domestic space heating fuel cycle.

Contrary to the fossil fuels-to-electricity fuel cycles the $SO_2$ and $NO_x$ emissions from the upstream process steps taken together are of the same order of magnitude as the emissions from the boilers themselves. Besides for the greenhouse gases the site of the emission site is important for the resulting impact. While for the oil-to-domestic space heating fuel cycle about 80% of the $SO_2$ and $NO_x$ emissions occur in Western Europe, for domestic space heating from gas this percentage is much lower, only about 55–60%. A high percentage of the particulate emissions from the oil fuel cycle, about 60 %, occur in the GUS. The percentage of emissions outside of Europe and GUS for oil is considerable (about 15–25%) for all of the five pollutants shown. For gas none of the pollutants are emitted outside of Europe and GUS, however, the percentage of emissions in the GUS is high, especially for NMHC (about 85%). NMHC is mostly of interest as a precursor for ozone. As is explained in section 3.1.4, for ozone effects an average model applicable in Europe and the GUS is employed because a site-specific ozone model is not available. So, for NMHC a site-specific analysis is not possible anyway.

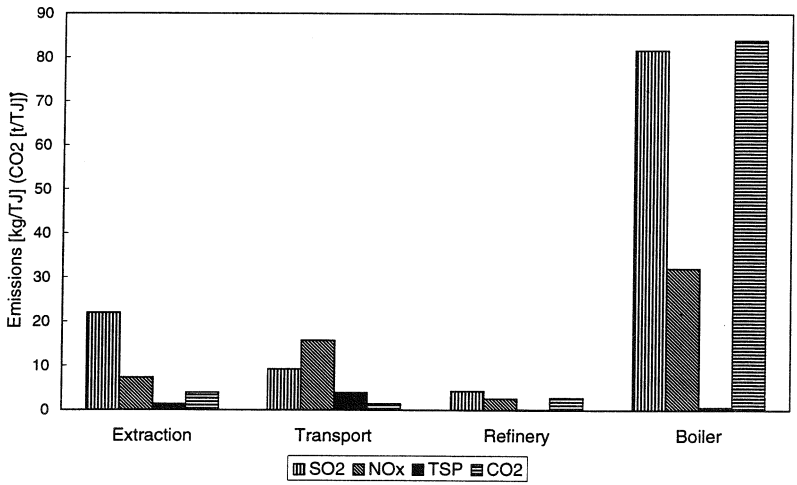



**Figure 3.6** Emissions of boiler with atomising oil burner and its upstream process steps

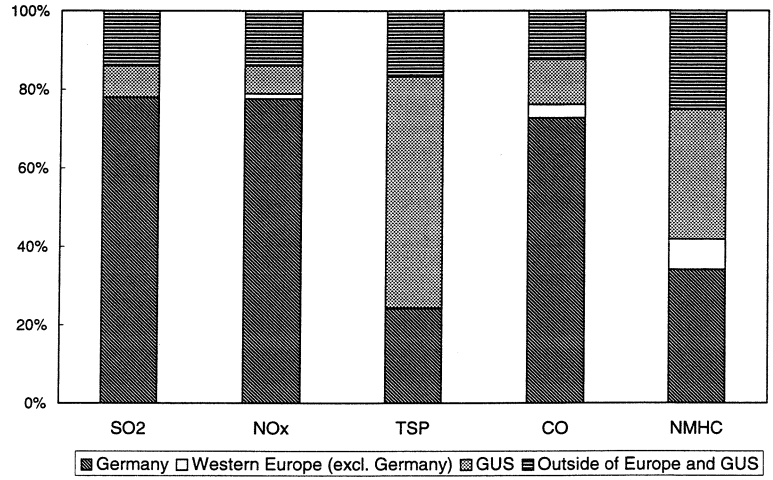



**Figure 3.7** Regional distribution of emissions of oil-to-domestic space heating fuel cycle

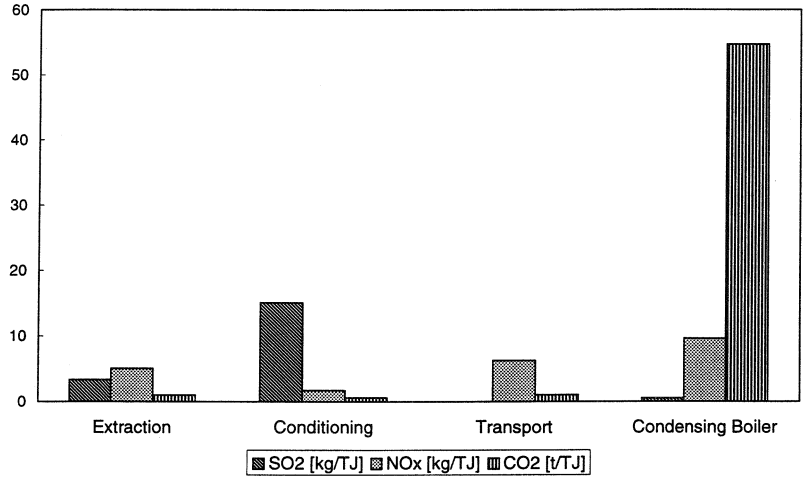



**Figure 3.8** Emissions of gas-fired condensing boiler and its upstream process steps

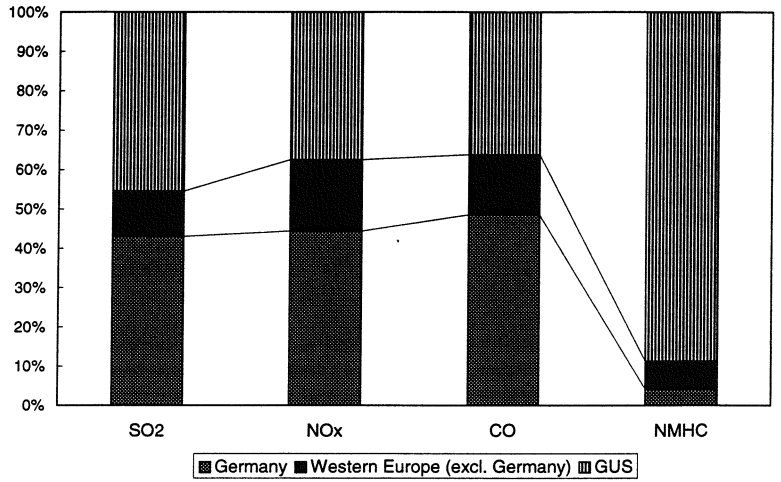



**Figure 3.9** Regional distribution of emissions of gas-to-domestic space heating fuel cycle

***Reference Technology for Domestic Space Heating with Electricity***

In 1992 8% of all flats were electrically heated in Germany (Eckerle *et al.*, 1995). In West Germany more than 95% of the electrically heated flats are heated by thermal storage heaters, most of them decentrally (see Table 3.4). Electric domestic space heating systems are mostly employed in older buildings (75%), only 25% of them can be found in new buildings (Borstelmann, Rohne, 1993).

Thermal storage heating is resistance heating. Thereby, during low load of the electricity supply system electric energy is transformed to heat that is stored for later use. Electric thermal storage heating has been strongly supported by the electric supply companies in the last decades as it allows to level the nightly decrease of the load. At certain times that are set by the electric supply company (e.g. 8 hours in the night or 6 hours in the night and 4 hours in the afternoon) the thermal storage heaters of the consumers receive electric energy at cheaper rates.

Here, decentral thermal storage heaters are chosen as reference technology. The capacity of each appliance is in the range of 1 to 3 kW (Pfitzner, Schäfer, 1994). Because of the decentral installation in the rooms distribution losses are not incurred. While all of the up-taken final energy is released as heat it is not always done at the desired time. Thus, the annual utilisation ratio is only 98.5%.

**Table 3.4** Percentages of electrically heated flats in West Germany in 1990

| | | |
|---|---|---|
| Direct heating | 1.8% | |
| Thermal storage heating | 95.0% | |
| • decentral thermal storage heater | | 90.8% |
| • thermal storage floor heating | | 2.9% |
| • central thermal storage heater | | 1.9% |
| Heat pump | 2.6% | |

Source: Borstelmann, Rohne, 1993

**Table 3.5** Electricity supply mix for thermal storage heating in Germany

| Electric supply company | Percentage of German electricity supply | Water | Nuclear | Lignite | Coal | Oil | Gas |
|---|---|---|---|---|---|---|---|
| Badenwerk | 10.5% | 0% | 65% | 12% | 23% | 0% | 0% |
| Bayernwerk | 14.0% | 0% | 55% | 13% | 32% | 0% | 0% |
| VEW | 5.0% | 0% | 1% | 0% | 91% | 0% | 8% |
| RWE | 30.0% | 0% | 50% | 1% | 47% | 0% | 2% |
| Neckarwerke | 1.0% | 0% | 64% | 0% | 36% | 0% | 0% |
| Preussen Elektra | 23.0% | 0% | 0% | 0% | 36% | 0% | 2% |
| HEW | 0.5% | 0% | 82% | 18% | 0% | 0% | 0% |
| VEAG | 16.0% | 0% | 0% | 97.5% | 1.5% | 1% | 0% |
| *Average (VDEW1.0)* | | *0%* | *36%* | *17%* | *46%* | *0.2%* | *0.8%* |
| GEMIS2.1 | | 0% | 15% | 15% | 70% | 0% | 0% |

Source: FDE (1996)

**Table 3.6** Emissions [kg/TJ useful energy] ($CO_2$ [t/TJ useful energy]) of the electricity supply mix of thermal storage heating

| Pollutant | Average | Baden-werk | Bayern-werk | VEW | RWE | Neckar-werke | Preussen Elektra | HEW | VEAG |
|---|---|---|---|---|---|---|---|---|---|
| $SO_2$ | 167.0 | 97.7 | 122.2 | 256.8 | 143.6 | 113.7 | 254.0 | 45.2 | 223.5 |
| $NO_x$ | 172.6 | 107.4 | 130.6 | 253.0 | 143.6 | 112.8 | 241.7 | 67.2 | 209.2 |
| TSP | 22.8 | 15.0 | 17.9 | 32.4 | 19.7 | 16.3 | 32.1 | 9.7 | 25.3 |
| $CO_2$ | 187.0 | 108.4 | 135.7 | 264.9 | 141.2 | 105.3 | 252.7 | 66.9 | 295.2 |
| CO | 107.7 | 97.3 | 100.6 | 121.9 | 102.2 | 96.5 | 115.4 | 92.7 | 125.2 |
| $CH_4$ | 660.7 | 344.1 | 466.5 | 1302.7 | 678.1 | 519.7 | 1262.4 | 32.1 | 59.0 |
| NMHC | 8.7 | 5.5 | 6.6 | 14.7 | 8.3 | 6.6 | 13.4 | 2.9 | 6.8 |
| $N_2O$ | 8.3 | 4.6 | 6.0 | 13.3 | 7.1 | 5.4 | 12.9 | 2.0 | 9.0 |

Source: VDEW database 1.0

Fichtner Development Engineering (FDE) have analysed for the VDEW1.0 database the electricity supply mix for thermal storage heating from data of the different German electric supply companies (see Table 3.5). On average thermal storage heaters are mostly supplied by

electricity from coal (46%), followed by nuclear energy (36%) and lignite (17%). Oil and gas account for only minor parts in this mix. This result constitutes a change to former studies (GEMIS 2.1 (Fritsche *et al.*, 1994), Konstantinidis *et al.* (1986), Friedrich *et al.* (1985)) that had assumed a much higher percentage of coal. However, as Table 3.5 shows, the percentages vary considerably between the different electric supply companies.

Table 3.6 lists the emission factors calculated with the VDEW1.0 database for the different electricity supply mixes of Table 3.5. The emission factors for the average mix are considerably higher than for the two boiler heating systems. The emission factors are lower when nuclear contributes considerably to the mix, but only if the nuclear percentage reaches 82% (HEW) the emissions are lower than for the oil-fired reference system, but not as low as for the gas-fired one.

### 3.1.3 The Priority Impact Pathways

For the two fossil-fired boilers the same impact pathways are important as for the other fossil energy systems (European Commission, 1995a):

- effects of atmospheric pollution on
  - human health,
  - crops,
  - forests and unmanaged ecosystems,
  - materials
- occupational health effects,
- effects due to oil spills, and
- impacts of global warming.

No new priority impact pathways could be identified.

For electric domestic space heating all the impact pathways identified for the fossil and nuclear energy systems are relevant, depending on the respective electric supply mix. In (European Commission, 1995b) for nuclear the following impact pathways were identified:

- effects of radioactive substances (emitted during normal operation of all process steps or due to an accident in the power plant) on human health, and
- occupational health effects.

### 3.1.4 Methodological Issues – Air quality modeling for low annual emissions and low stack heights

The question was whether the two air quality models implemented in EcoSense were appropriate for a marginal assessment of the concentration and deposition increments due to the emissions of the two fossil-fired boilers. The annual emission loads of the two boilers are about 1 to 10 kg $SO_2$ or $NO_x$ per year while due to the calculation precision of the (regional) Harwell Trajectory Model pollution increments can be only assessed for annual emission loads above about 500 t $SO_2$ or $NO_x$. Therefore, the emissions of the boilers have been scaled up by several orders of magnitude. Test runs have shown that up to annual emission loads of about 2000 t the results prove to be about proportional to the emission loads, so that the re-

sults are not distorted by the scaling. Above 2500 t, the results are no longer linear and the ratio between $SO_2$ and $NO_x$ influences the increments considerably. The annual emission load is not a problem for the Gaussian plume model employed for the local range analysis, but due to the lower stack height, the uncertainty of the local results increase, especially if there is a 'rough' topography.

### 3.1.5 The Quantification of Damage Costs

***Quantification for the fossil-fired heating systems***

The impacts and damages of the two fossil-to-domestic space heating fuel cycles under analysis were quantified following the EXTERNE methodology established by now. Appendix I to VII summarise the approaches, exposure-response functions and monetary values used and describe the software tool EcoSense employed for the assessment.

In Table 3.7 the quantified damages for the two fuel cycles are summarised. As for the fossil power plants, the damage costs due to air emissions of the upstream process steps of the boilers have been assessed by assuming that the emissions occur at the site of the boiler. Employing the YOLL approach for mortality effects without global warming, for the oil reference system damage costs of 2.2 mECU/MJ were quantified, employing the VSL approach the damages increase to 6.8 mECU/MJ. The respective results for the gas system are 0.7 mECU/MJ and 2.5 mECU/MJ.

Increased mortality, mostly due to sulfate and nitrate aerosols, contributes the most to the total quantified damage costs (without global warming). As these pollutants are the result of chemical transformations from the $SO_2$ and $NO_x$ directly emitted by the reference systems, the share of the local range of the total damage costs is only small. Thus, the total quantified damage costs are not affected by the increased uncertainty of the Gaussian plume model results compared to the power plants (section 3.1.4).

As site-specific ozone models are not available at the moment, ozone impacts are assessed using the damage costs derived by Rabl and Eyre (1997) with a simplified approach. In this approach results for the ozone models of EMEP and Harwell were combined with the exposure-response functions and economic valuation for health and agricultural crops recommended in (European Commission, 1995) to derive average damage factors for $NO_x$, NMHC and $CH_4$ emissions in Europe (incl. GUS).

Using the IPCC range of 1.1 to 27.1 ECU/t $CO_2$ for the global warming damages, damage costs of 0.1–2.5 mECU/MJ due to the $CO_2$, $CH_4$ and $N_2O$ emissions of the oil reference energy system have been quantified; 0.07–1.8 mECU/MJ for the emissions of the gas reference energy system. With the EXTERNE results (illustrative restricted range) for global warming (see Appendix V) the damages are higher: 1.7–4.3 mECU/MJ and 1.2–3.0 mECU/MJ, respectively. The present results for global warming are about the same order of magnitude as all the other damage costs quantified together, the uncertainty of the global warming damages, however, is especially high. Table 3.8 summarises the totals of the quantified damages, de-

pending on the choice of YOLL/VSL for mortality effects and on the global warming damage costs per tonne $CO_2$.

Table 3.9 shows the totals of the quantified damages related to the fuel input instead of the useful energy produced. For the oil reference energy systems damage costs of 0.1–0.6 ECU/l oil input (YOLL approach) have been quantified. Eckerle *et al.* (1995) list a household price for domestic fuel oil of about 0.24 ECU/l oil (in 1993 prices), i.e. the quantified damage costs are about the same order of magnitude as the present costs for the consumers. For gas this pattern is less marked. The fuel-specific damage costs for the gas reference energy system are 0.03–0.1 ECU/$m^3$ gas input, the household price is about 0.3 ECU/$m^3$ gas.

***Quantification for electric thermal storage heating***

There are two reasons why for the electric thermal storage heating system a true marginal analysis cannot be carried out:

- About all emissions and impacts are caused by the upstream process steps, i.e. the electricity generation in various power plants distributed across Germany (and Europe). Additional facilities are not installed.
- It is not possible to specify the sites of the emissions, not even roughly as it has been possible e.g. for the German photovoltaics and wind reference energy systems (Krewitt *et al*, 1997).

Electric heating is not the only reference energy systems affected by these aspects. The analysis of photovoltaics and wind as well as the rational use of energy (Hohmeyer *et al.*, 1996) faces similar problems.

As by the installation of one additional electric thermal storage heater the emissions are increased all over Germany by a slight amount, an average approach is justified for the impact assessment. Krewitt and Heck (1997) have quantified the damage costs of the electricity production in Germany in 1990 and derived average damage costs per tonne pollutant from it. Thereby, the average damage costs differ considerably from the costs derived for the fossil power plants in Lauffen from the German Implementation of the EXTERNE project (Krewitt *et al.*, 1997). These differences are caused by the spatial variation in $SO_2$, $NO_x$ and $NH_3$ emissions across Germany. For example, the damage costs per tonne of $SO_2$ emitted by the Lauffen plants are higher than the damage costs derived for the whole electricity production because Lauffen is located in a region with relatively low $SO_2$ background emissions and enough $NH_3$ available, which causes the formation of additional ammonium sulfate.

**Table 3.7** Damages costs [mECU/MJ] of the oil- and gas-fired reference heating system

| | Boiler with atomising oil burner (blue burner) | Gas-fired condensing boiler | $\sigma_g$ |
|---|---|---|---|
| *POWER GENERATION* | | | |
| Public health | | | |
| Mortality – YOLL[1] *(VSL)* | 1.2 (5.0) | 0.1 (0.5) | B |
| *of which* *TSP* | 0.01 (0.03) | 0 (0) | |
| *SO2* | 0.8 (3.7) | -0.0001 (0.003) | |
| *$NO_x$* | 0.3 (1.2) | 0.1 (0.5) | |
| Morbidity | 0.1 | 0.0 | A |
| Accidents | n.q. | nq | A |
| Occupational health (see below) | | | A |
| Ozone impacts (human health & crops) – YOLL[1] *(VSL)* | 0.1 (0.4) | 0.01 (0.1) | |
| Crops | -0.001 | 0 | B |
| Ecosystems | iq | iq | B |
| Materials | 0.02 | 0.001 | B |
| *Monuments* | | | |
| Noise | ng | ng | |
| Visual impacts | ng | ng | |
| Global warming[2] | | | C |
| Conservative 95 % confidence interval | 0.3–11.7 | 0.2–7.6 | |
| Illustrative restricted range | 1.5–3.9 | 1.0–2.5 | |
| *OTHER FUEL CYCLE STAGES* | | | |
| Public health – YOLL[1] *(VSL)* | 0.8 (0.9) | 0.4 (1.6) | |
| *of which* *TSP* | 0.1 (0.3) | 0 (0) | |
| *$SO_2$* | 0.4 (1.6) | -0.0001 (0.003) | |
| *$NO_x$* | 0.3 (1.1) | 0.1 (0.5) | |
| Occupational health (incl. power generation) | 0.03 | 0.002 | |
| Ozone impacts (human health & crops) – YOLL[1] *(VSL)* | 0.05 (0.4) | 0.05-0.08 (0.18-0.25) | |
| Crops | 0.000 | -0.0004 | B |
| Ecological effects | nq | nq | B |
| Materials | 0.01 | 0.01 | B |
| Road damages | ng | ng | A |
| Oil spills | 0.002 | - | |
| Global warming[2] | | | C |
| Conservative 95 % confidence interval | 0.04–1.3 | 0.04–1.5 | |
| Illustrative restricted range | 0.2–0.4 | 0.2–0.5 | |

[1] Yoll= mortality impacts based on 'years of life lost' approach; here results are calculated assuming a discount rate of 3%, VSL= impacts evaluated based on 'value of statistical life' approach.

[2] $CO_2$ equivalent emissions are calculated from the $CH_4$ and $N_2O$ emissions employing the Global Warming Potentials of IPCC (1996a) for the time horizon of 100 years.

ng: negligible; nq: not quantified; iq: only impact quantified; - : not relevant

**Table 3.8** Sub-total damages of the oil- and gas-fired reference heating systems and the electric thermal storage heating system (average electricity supply mix)

| | Boiler with atomising oil burner (blue burner) | Gas-fired condensing boiler | Electric thermal storage heating |
|---|---|---|---|
| YOLL *(VSL)* | | | |
| Conservative 95 % confidence interval | 2.6–15.2 (7.2–19.8) | 0.9–9.8 (2.7–11.6) | 3.6–31.1 |
| Illustrative restricted range | 3.9–6.5 (8.5–11.1) | 1.9–3.7 (3.7–5.5) | 6.5–12.2 |

**Table 3.9** Damage costs of the oil-fired and gas-fired heating systems relative to supplied fuel

| | Boiler with atomising oil burner (blue burner) [ECU/l oil] | Gas-fired condensing boiler [ECU/$m^3$ gas] |
|---|---|---|
| FUND - YOLL *(VSL)* | | |
| Conservative 95 % confidence interval | 0.1–0.6 *(0.3–0.8)* | 0.03–0.4 *(0.1–0.4)* |
| Illustrative restricted range | 0.2–0.3 *(0.3–0.5)* | 0.07–0.1 *(0.1–0.2)* |

**Table 3.10** Damages by pollutant

| | ECU / t of pollutant |
|---|---|
| $SO_2$ *- YOLL *(VSL)* | 11200 *(46700)* |
| $NO_x$ *- YOLL *(VSL)* | 11900 *(39700)* |
| $PM_{10}$ *- YOLL *(VSL)* | 16900 *(57200)* |
| $NO_x$ (via ozone) | 1500 *(11300)* |
| $CO_2$ | |
| conservative 95 % confidence interval | 3.8–139 |
| illustrative restricted range | 18–46 |

* Yoll= mortality impacts based on 'years of life lost' approach; here results are calculated assuming a discount rate of 3%, VSL= impacts evaluated based on 'value of statistical life' approach.

**Table 3.11** Damages [mECU/MJ useful energy] for the electric heating system (average electricity supply mix)

| | | |
|---|---|---|
| Public health | | |
| Mortality – YOLL[1] | | 2.1 |
| *of which* | *TSP* | 0.3 |
| | $SO_2$ | 1.2 |
| | $NO_x$ | 0.6 |
| Morbidity | | 0.3 |
| Accidents | | n.q. |
| Occupational health | | 0.07 |
| Ozone impacts (human health & crops) – YOLL[1] | | 0.4 |
| Crops | | 0.02 |
| Ecosystems | | i.q. |
| Materials | | 0.04 |
| *Monuments* | | n.q. |
| Noise | | ng |
| Visual impacts | | ng |
| Oil spills | | 0.00004 |
| Damages due to nuclear electricity | | 0.0005 |
| Global warming[2] | | |
| Conservative 95 % confidence interval | | 0.8-28.3 |
| Illustrative restricted range | | 3.7-9.4 |

[1] Yoll= mortality impacts based on 'years of life lost' approach; here results are calculated assuming a discount rate of 3%, VSL= impacts evaluated based on 'value of statistical life' approach.

[2] $CO_2$ equivalent emissions are calculated from the $CH_4$ and $N_2O$ emissions employing the Global Warming Potentials of IPCC (1996a) for the time horizon of 100 years.

ng: negligible; nq: not quantified; iq: only impact quantified; - : not relevant

The ozone impacts have been quantified with the damage factors derived by Rabl and Eyre (1997) (see above). Occupational health impacts, oil spill damages and damage due to the use of nuclear energy have been calculated from the National Implementation results under consideration of the different net efficiencies of the (fossil) power plants employed. Table 3.11 summarises the quantified damages for the electric reference heating system for the average electricity supply mix. With the EXTERNE results (illustrative restricted range) for global warming, damage costs of about 4–9 mECU/MJ are quantified, which is about the same order of magnitude as all the other damage costs quantified together, the uncertainty of the global warming damages, however, is especially high. Table 3.8 summarises the totals of the quantified damages, depending on the choice of YOLL/VSL for mortality effects and on the global warming damage costs per tonne $CO_2$.

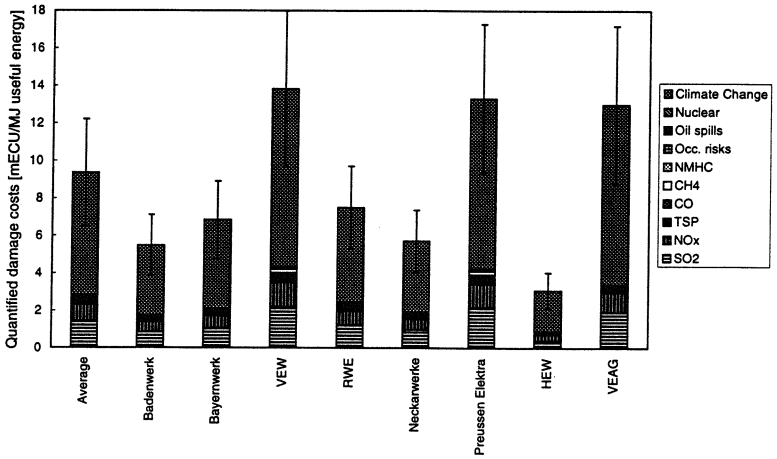



**Figure 3.10** Quantified damage costs [mECU/MJ useful energy] for the thermal storage heating system (mid value of illustrative restricted range for global warming, error bars indicate illustrative restricted range) – Differences between electric supply company due to differences in electricity supply mix

Table 3.8 gives the sub-total of the quantified damages, depending on the choice of YOLL/VSL for mortality effects and on the global warming damage costs per tonne $CO_2$. However, the results vary substantially depending on the electric supply mix (see Figure 3.10); the higher the share of nuclear energy in the supply mix is, the lower the quantified damage costs are. But even for the option with the highest share of nuclear energy higher damage costs are quantified than for the two fossil-fired reference heating systems.

### 3.1.6 Conclusions

It was the objective of this sub-task to extend the methodology for the assessment of external costs developed so far in the EXTERNE project to the analysis of external costs of domestic space heating. Thereby, the focus of the analysis was on central space heating with oil and gas and electric thermal storage heating. For the analysis a certain heating task was defined and the quantity 'useful energy produced' was used as functional unit. Thus, the quantified external costs depend only on 'technical' factors while factors related to the weather or to consumer behaviour do not influence the result.

In this study as heating task the central space heating of an one-family house, located in Lauffen, with a flow temperature of 70 °C and full load hours of 1650 h per year was defined. Modern heating systems with the present state of technology were taken as reference technologies. For oil this is a low-temperature boiler with an atomising oil burner (blue-burner)

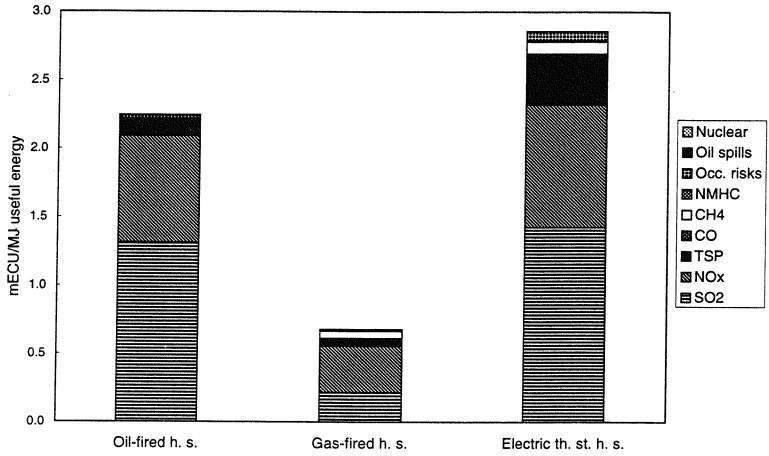



**Figure 3.11** Distribution of the quantified damage costs – excl. global warming – on the different pollutants and burdens

and for gas a condensing boiler. Decentral thermal storage heaters were chosen as reference technology for electric space heating. The emissions of the upstream processes of the reference energy systems were quantified based on the GEMIS 2.1 model employing the VDEW 1.0 database. Thereby, for the electric heating system the electricity supply mix was based on data of the different German electric supply companies. On average thermal storage heaters are mostly supplied by electricity from coal (46%), followed by nuclear energy (36%) and lignite (17%). Oil and gas account for only minor parts in this mix.

The impacts and damages of the two fossil-to-domestic space heating fuel cycles under analysis were quantified following the EXTERNE methodology established by now. An analysis showed that the available air quality models can be used without introducing a major error into the results, even though the additional annual emission loads of the heating systems are small and the stack heights low. The $SO_2$ and $NO_x$ emissions from the upstream process steps of the oil- and gas-fired boilers are of the same order of magnitude as the emissions from the boilers themselves, in this the fossil reference heating systems differ from the fossil electricity fuel cycles. However, as has been demonstrated in the analysis, this fact does not affect the quantified damages in total, so that the EXTERNE methodology developed for fossil fuel to electricity fuel cycles can be applied without further changes.

As by the installation of one additional electric thermal storage heater the emissions are increased all over Germany by a slight amount, an average approach is justified and even more appropriate for the impact assessment. Thus, damages were quantified employing the damage

costs per tonne pollutant derived as part of the aggregation task of the German Implementation of EXTERNE.

Increased mortality, mostly due to sulfate and nitrate aerosols, contributes the most to the total quantified damage costs (without global warming). The present results for global warming are about the same order of magnitude as all the other damage costs quantified together, the uncertainty of the global warming damages, however, is especially high. Using the so-called 'illustrative restricted range' EXTERNE results for global warming and the YOLL-approach for the valuation of mortality impacts, the following costs were quantified:

- for oil: 3.9–6.5 mECU/MJ,
- for gas: 1.9–3.7 mECU/MJ, and
- for electric (average electric supply mix): 6.5–12.2 mECU/MJ.

For electric thermal storage heating, not surprisingly, the results vary substantially depending on the electric supply mix; the higher the share of nuclear energy in the supply mix is, the lower the quantified damage costs are. But even for the option with the highest share of nuclear energy higher damage costs are quantified than for the two fossil-fired reference heating systems.

For the oil reference energy systems damage costs of 0.1–0.6 ECU/l oil input (YOLL approach) have been quantified, i.e. the quantified damage costs are about the same order of magnitude as the present costs for the consumers. For gas this pattern is less marked. The fuel-specific damage costs for the gas reference energy system are 0.03–0.1 ECU/$m^3$ gas input, the household price is about 0.3 ECU/$m^3$ gas.

### 3.1.7 References

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## 3.2 Demand Side Options

### 3.2.1 General methodology and reference technologies

***General methodology***

In general, the impact assessment in this report follows the methodology of the impact assessment of end-use technologies suggested in Hohmeyer *et al.* (1996). External costs of these technologies basically are due to production, installation, and disposal. The use of these technologies usually induces only minor impacts. The costs of the important phases in the life-cycle of end-use technologies can be related to a certain amount of energy saved by initially introducing end-use technologies or by replacing standard/conventional technologies by advanced energy efficient technologies. In the case of replacing less energy efficient technologies with advanced new devices, the incremental impact of the new compared to the conventional technology is the basis for external cost calculations. According to the proposed methodology, the last production step (usually: final assembly) of the considered technologies has to be assessed by interviews e.g. with manufacturers. The burdens of upstream (pre-production) processes can be calculated with EMI 2.0 - an input-output-model that has been extended by sector specific emission-coefficients for the German economy split up into 58 production and service sectors. In this study the model has been run using the input-output structure of the German economy for the base year 1990. The technologies examined were:

- refrigerators,
- windows, and
- heat exchangers.

Heat exchangers are treated as a new technology not directly replacing an old one. The replacement of energy production facilities due to energy savings are not taken into account. Hence, the total impacts of production, installation and disposal are the basis for external cost accounting. In contrast, for refrigerators and windows only the incremental impact of substituting a conventional by an advanced technology is taken into account.

***Refrigerators***

*Basic principles*
About 12% of the total electricity demand of private households (excluding electric heating of air and water) is used for refrigeration purposes (VDEW, 1992). Hence, an efficient use of electricity by refrigerators seems to have a large potential for energy saving. The production of a refrigerator requires about 1100 kWh of primary energy per appliance (RAVEL, 1992) which compares to an energy-consumption between 1600 - 8200 kWh over the course of a mean lifetime of 15 years. As the most energy-efficient refrigerators consume only about 2/3 of the energy demanded by standard refrigerators, a higher energy-requirement in the production of energy-efficient refrigerators can be positively balanced by the energy-savings during the lifetime. The principle of cooling is the same for almost all refrigerators. A compressor condenses the cooling substance which is then liquefied in the condensator. The resulting la-

tent heat is led off to the outside air. In the evaporator the cooling substance transforms back from the liqueous to the gaseous phase cooling down the direct environment. Then the cycle starts again. The ratio between the heat withdrawn from the appliance and the electric energy input ranges between 0.7 and 1.2 for the refrigerators on the market in 1991. For the future, values of about 2 are predicted (Deutscher Bundestag, 1990). Refrigerators exist with and without an evaporation compartment. The evaporation compartment can be held on 0°C (zero stars), -6°C (*), -12°C (**) and -18°C (***). In Germany only ***-compartments are common. A refrigerator with an evaporation compartment always consumes more electricity than one without. Both, refrigerators with and without an evaporation compartment will be examined.

*Production, installation, and disposal of refrigerators*
Most of the impacts during the lifetime of a refrigerator are due to its production, installation, and disposal. ZEW tried hard to obtain respective figures from the folowing German manufacturers: AEG Hausgeräte AG, Bauknecht Hausgeräte GmbH, Robert Bosch Hausgeräte GmbH, Liebherr Hausgeräte GmbH and Whirlpool Hausgeräte Vertriebs GmbH. Unfortunately, none of them wanted to co-operate with ZEW. Most of them were either too busy with eco-auditing procedures and felt that they did not have the time to gather the required data in their company, or that the data was too confidential to be distributed outside the company.

Originally it was planned to compare the impacts due to production, installation, operation and disposal of a "best selling" refrigerator with those of "a most efficient" cooling device. The unwillingness of the manufacturers to co-operate forced ZEW to simplify the methodology. We now compare "a most efficient " (energy saving category A) with "a less efficient refrigerator " (category B). We assume that production, installation and disposal are basically the same for both types except for the use of insulation material (EPS foam). Even the final assembly process is assumed to be identical. However, we compare identical cooling and freezing volumes, i.e. "the most efficient" refrigerator needed to be bigger due to increases in insulation volume. Accordingly, the "most efficient" refrigerator needed more coating material (sheet metal and varnish) to provide the same cooling and freezing services. However, we felt that the impacts due to the production of additional sheet metal and varnish would be small compared to the impacts of additional insulation material so that they are not included in the impact calculation. Accordingly, the calculated impacts of efficient refrigerators will be a bit too low. The incremental impact hence is assumed to be solely due to the additional production of EPS foam.

***Windows***

About 75% of the total energy consumption of private households in Germany are used for space heating. As a result the German Thermal Insulation Ordinance (GTIO) sets high standards for insulating measures which have to be fulfilled when constructing a new or renovating an old building. This study compares the external costs of a triple glassed window with that of a window with two panes. The relevant data for the impact assessment has been generated in 1985 within the context of an EU-Project (Contract-No. XVII/AR/83/289) titled "Employment effects of energy conservation investments in EC countries" conducted by

Hohmeyer *et al.* (1985). In this study the additional inputs that are necessary for the production and installation of a triple-glassed window instead of a conventional frame with two window panes were examined.

*Basic principle*
The insulation potential of a material is expressed by its k-value. The k-value stands for the thermal energy transfer through the cross-section of a material at a given temperature difference between the inside and the outside: k = [W/(m$^2$K)]. The energy of the sun which enters the window at a perpendicular incidence is characterized by the g-value and it corresponds to the transmission ratio. A higher value means more sun-energy flux through the window what induces a higher rate of passive heating of the building. Therefore it is important not to look exclusively at the k-value. Currently, windows have the highest heat transfer coefficient k of the house envelope, i.e. a relatively big amount of heating energy is lost through the window surface. Installing windows with a superior technology is therefore the most effective way of lowering the k-value of the entire building. The thermal insulation effect of windows varies due to several factors:

- number of layers,
  adding a second layer will lower the k-value of the window by a factor of two
- distance between the panes,
  due to the strong thermal insulation of stationary air; the ideal distance between the panes is 12 to 16 mm
- kind of gas between the panes,
  the space between the panes is filled with inert gases, such as argon or krypton, which have a thermal conduction coefficient lower than that of air
- surface treatment of the panes,
  the indoor surface of the panes is sometimes coated with a metallic layer, which allows sunlight (short wave radiation) in from the outside, but which can hardly be penetrated by heat radiation (long wave radiation) from the inside, creating a green-house effect

The German window market offers several kinds of windows with differing thermal insulation properties. Table 3.12 below lists the most important types of windows that are presently available on the market. The first two represent the kinds which are most common where the second just fulfils the requirements of the German Thermal Insulation Ordinance (GTIO) of 1995 and is nowadays most often installed. The second part of the table (3...7) shows the alternatives which surpass the GTIO.

*Production, installation and disposal of windows*
The main components of a window are the panes (float glass), the window frame and the joints. The window frame can be made of wood, poly-vinyl-chloride (PVC) or aluminium. In terms of the external costs of production, the different frame materials vary. Wood seems to have the least external costs as it is a natural, renewable resource. However the substances used to preserve and protect the wood against fungus and weather pose problems. Plastic windows consist of about 70-75% PVC, a product of the chemical (chlorine-) industry. The remaining 25 - 30% are various modifiers, stabilisers, colour-pigments and other aids. The

**Table 3.12** Window types currently available on the German market

| Type of window | k-value ($W/m^2/K$) | g-value |
|---|---|---|
| 1. standard two layer pane | 2.6 | 0.77 |
| 2. standard two layer thermal insulation | 1.4 | 0.69 |
| 3. three layer thermal insulation pane with Argon filling | 0.78 | 0.52 |
| 4. three layer thermal insulation pane (Krypton filling) | 0.65 | 0.52 |
| 5. two layer compound glass with selective coating (air) | 1.1 | 0.54 |
| 6. two layer compound glass with both side selective coating and Argon-filling | 0.65 | 0.48 |
| 7. two layer glass with multiple film and air-filling | 0.6 | 0.4 |

Source: IWU (1995)

waste disposal of plastic windows may represent a hazard. Especially the additives represent a problem as they can impede the recycling of the plastic as a whole. The production of aluminium is very energy intensive. The window pane consists of mineral-oxides which are processed by the glass-industry. In order to get the k-value of the window pane below k = 1.1 the windows have to be coated by a metallic layer, e.g. a silver-alloy.

The single components from the different specialised manufacturers are delivered to the window maker who assembles the windows. In order to increase their thermal insulation power, the assembly of the windows also comprises coating the indoor surface of the panes with a metallic layer, and filling the space between the panes filled with Argon or Krypton.

The installation of a window involves the following steps:
- fixing the window to the masonry
- filling the gap between the masonry and the window with foam
- roughcasting the masonry around the window

The maintenance requirements of a window are influenced by the weather conditions it is exposed to. The maintenance concerns mainly the frame material and the rubber sealing. While plastic frames need no maintenance wooden frames have to be recoated with protection solvents regularly. In this study the substitution of frame materials is not examined. Regarding the comparison of the insulation properties of windows - inquired in this study - the maintenance of the different types of insulation windows can be expected to be the same. Regarding the time of operation therefore only the heat loss through the surface of the window might be of relevance for the calculation of the external costs.

The lifetime of a window is usually expected to be between 25 and 50 years. Then the window has to be disposed, burnt or recycled. Table 3.13 shows the potential of disposal, recycling and burning of window residues. The values are taken from a life-cycle study by Richter *et al.* (1992) and constitute a forecast of the recycling-situation.

**Table 3.13** Potential of disposal, recycling, and burning of window residues

| | Burning | Recycling | Disposal |
|---|---|---|---|
| Wood | 80% | - | 20% |
| Aluminium | 30% | 50% | 20% |
| Steel | 40% | 30% | 30% |
| PVC | 40% | 30% | 30% |
| other plastics | 80% | - | 20% |

Source: Richter *et al.* (1992)

Concerning the calculation of external costs, the different frame materials show different behaviours. While the production of primary aluminium has the highest primary energy intake of all the named frame materials, the processing of secondary (recycled) aluminium takes only 5 - 10% of the energy of primary aluminium. The variety of additives used in the production of plastic frames impedes the recycling of PVC. As plastic windows contain heavy metals, deposition is an environmental burden which can represent an external effect. Besides $CO_2$ and CO, burning of PVC sets free hydrochloric acid, which cannot be totally removed by a fluegas precipitator. On the other hand energy can be regained by burning the waste in a CHP plant. The latter is especially of interest for wood. Burning of the wooden frame in a CHP plant considerably improves the total energy-balance of the window.

***Heat exchangers***

The calculation of external costs due to the production, installation, use and disposal of heat exchangers is based on input-output-analysis. The results of this study are closely linked to a German study on employment effects of demand-side technologies for the German EPA (Umweltbundesamt), which has been carried out in 1992/93 under the responsibility of the German Institute for Economy (DIW). In that study (Umweltbundesamt, 1993) input-output-analysis was applied to assess employment impacts. The use of heat exchangers in the manufacturing sector have been dealt with in a special case study generating an input-output structure for the production of heat exchangers, the extension of heat exchanger production capacity and for the maintenance of the production facilities as an intermediate result. Fortunately, it is exactly the kind of structure that is necessary for the of use the EMI 2.0 model. Accordingly, the input structure for heat exchangers has been taken from the DIW-study.

*Basic principle*
Vitiated air (waste heat), waste gases, and cooling water from industrial processes usually have higher temperature levels than the fresh air and water used. Regaining waste heat from industrial processes via heatexchangers is considered to be the most economic and effective energy saving measure in the manufacturing sector (Cube, 1983: 103) cited in (Umweltbundesamt, 1993: 20-2).

Heat exchangers can be divided into heat exchangers for gases and for fluids. Due to the higher heat capacity of its medium, the latter is more compact. Heat exchangers for vitiated gases can be divided into:

- heat diffusion between fresh air and vitiated gases along a mutual partition which could be plates or tubes - the vitiated and fresh air streams therefore have to be close to each other
- heat transmission with a secondary fluid or gas cycle where the contact surfaces are usually tubes - this sort of heat exchanger is common when the ingoing and outgoing streams are not close to each other
- heat transmission via a rotating heat carrier which takes off the heat of the vitiated air and then circulates through the fresh air stream and gives off the heat - the heat carrier is usually solid or a fluid

There are two kinds of heat exchangers that regain heat from fluids:

- heat transmission with bundled tubes - one fluid flows through the tubes and the second fluid is flowing around the tubes
- heat transmission by means of plates - the heat spending and the heat taking fluids alternately flow through a sequence of plates which on one surface take off the heat and on the opposite side give off the heat to the heat taking fluid

### 3.2.2 Specific methodological issues

The specific methodological issues of end-use technologies have already been mentioned in Hohmeyer *et al.* (1996, p.25:).

1. The operation of these technologies normally causes no or extremely low emissions.
2. They do not demand (additional) fuel for their operation, reducing fuel consumption instead.
3. The major emissions due to the use of these technologies are caused through the direct and indirect economic activities induced by their production.
4. Their positive and negative impacts are overwhelmingly caused by existing technologies in the energy and industrial sectors not by additional new facilities.
5. The largest share of their negative effects can hardly be localised, as it is unknown in which exact location all the intermediate production takes place.

The specific meteorological data required by a Gaussian plume model for the site specific approach turned out to be rather expensive to obtain from meteorological service institutions. Hence, we decided to use the existing values for the Lauffen power plant by adjusting geographical longitude and latitude to the sites examined in this study. The average wind speed and the dominating wind direction of those sites turned out to be quite similar to the Lauffen values. Thus, we do not expect that actual site specific meteorological data would have led to substantially different results. We assume that the related uncertainty of the overall results (external costs per kWh) is rather low compared to the uncertainty of impact assessment and monetarisation. In addition to these well-known methodological issues, some new aspects occur in this study. Since the calculations with EMI2.0 provide emission data for the micropollutants arsenic, lead, mercury and cadmium, it is possible to calculate the impacts and external costs due to these pollutants in principle. As a first and rather simple approach, these micropollutants have been incorporated into the Industrial Source Complex Model of EcoS-

ense, i.e. only local impacts are accounted for. Furthermore, some exposure-response functions for arsenic and cadmium will be used to calculate the impact of the respective emissions on the additional occurrence of cancer. The related external costs will be presented under the subheading "Other impact pathways and external costs" for each technology examined. For mercury, WHO has published a threshold of 0.26 $\mu g/m^3$ and a background concentration of 10 $ng/m^3$. The exposure-response functions available to ZEW for arsenic are:

(As1) $$impact(cancer) = \Delta concentration \times population \times \frac{4}{7} \times 10^{-4}$$ (LAI)

(As2) $$impact(cancer) = \Delta concentration \times population \times \frac{2}{7} \times 10^{-5}$$ (US/EPA)

For cadmium the following exposure-response functions will be used:

(Cd1) $$impact(cancer) = \Delta concentration \times population \times \frac{12}{7} \times 10^{-2}$$ (LAI)

(Cd2) $$impact(cancer) = \Delta concentration \times population \times \frac{18}{7} \times 10^{-3}$$ (US/EPA)

The US/EPA unit risk factor will be used as the lower boundary for external costs, whereas the LAI unit risk factor will provide the upper value. For all of these functions no threshold will be assumed. The monetary valuation will be done according to the value of statistical human life as agreed upon in the ExternE which is ECU 2.6 million. Furhter, there is no damage function available for mercury. As a result, we only check if the impacts exceed the threshold value given by WHO. Finally, there will be no impact assessment done on lead due to a lack of exposure-response function for this micropollutant. It has to be pointed out that the emissions of these pollutants are caused by intermediate production. Thea are not specifically caused by the technologies analysed. If all other energy technologies in the ExternE project would be analysed for intermediate emission effects with a model like EMI2.0 similarexternal effects due to these pollutants would be found.

### 3.2.3 Quantification of impacts

***Impacts of energy efficient refrigerators***

*Burdens of energy efficient refrigerators calculated with EMI2.0*

We assume that the main impacts of production, installation, use, and disposal of energy efficient refrigerators compared to conventional cooling devices are due to the production and pre-production processes of the insulation material. Hence, we use the input-structure of EPS foam production to calculate the incremental emissions of a very efficient refrigerator (energy efficiency class A) compared to a less efficient device (energy efficiency class B). The structure of EPS foam production is given in Table 3.14. The given input-structure has been generated within the methodology study for end-use technologies in the scope of ExternE

(Hohmeyer *et al.* 1996). These coefficients will be multiplied by a certain monetary demand for refrigerators which will be determined within the scenario specified below.

*Scenario*
In Germany about 29.475 million refrigerators are in use. Table 3.15 shows their age structure as of 1991, subdivided into the primary and secondary device. 43% of the primary refrigerators of a household are younger than 6 years while 23% are older than 10 years. For secondary devices the age structure shifts towards older refrigerators. 47% of them are older than 10 years. The average lifetime of refrigerators is 15 years (Geiger *et al.* 1993).

In 1991, the German industry produced 4.226 million freezing and cooling units with a market volume of DM 2,273 million. Each year about 1.965 million refrigerators are sold in Germany. 45.7% of them include a ***-compartment. In the reference scenario we assume that in 15 years the total stock of 29.475 million existing units of existing refrigerators in Germany has to be replaced by devices of the energy efficiency class B. In the progressive scenario, we will replace all existing units with a refrigerator of the efficiency category A. To simplify calculations, we assume that all existing refrigerators are replaced in our base year 1990 and all energy saving are as well gained in 1990. The reference technologies are taken from a brochure of a manufacturer (AEG Hausgeräte AG). All chosen refrigerators have the same exterior dimensions, but the cooling compartments differ in volume. The class A refrigerator has less cooling volume than refrigerators of category B. We assume that the difference is due to the increasing use of insulation material in the class A-refrigerators.

We examine a refrigerator with (Scenario I) and one without (Scenario II) ***-compartment and artificially adjust them to compare the same cooling volumes. Table 3.16 gives the data necessary to calculate the additional demand for insulation material and the avoided demand of household electricity. The second and third column in the table give the data of the original class A technologies. Column six and seven show the adapted class A technologies with the same cooling volume as the less efficient devices given in column four and five. According to Table 3.16, the installation of a refrigerator (class A) with ***-compartment instead of a conventional device requires 13 litre additional insulation material and saves 35kWh electricity in private households per year. The installation of a class A refrigerator without ***-compartment requires 15 litre additional insulation material and saves 38kWh electricity in private households per year. With a total number of 29.475 million cooling units to be installed, the demand for EPS foam in the two scenarios amount to about 386.000$m^3$ (Scenario I) and 452.000 $m^3$ respectively. One $m^3$ of EPS foam costs about $DM_{90}$ 88.89 (Hohmeyer *et al.* (1996): 66, adjusted to $DM_{90}$ values). Thus the total monetary demand for insulation material in the scenarios amounts to $DM_{90}$ 34 million and $DM_{90}$ 40 million respectively. These demands multiplied by the input-coefficients given above provide the input-data necessary for the EMI2.0 calculations. The emissions due to EPS production are given in Table 3.17 below. In that table, burdens due to direct inputs are caused by the production of the inputs used in

**Table 3.14** Input coefficients of EPS foam production.

| No. | Sectors | Input-coefficients |
|---|---|---|
| 1 | Agricultural products | 0 |
| 2 | Forestry, fishery products | 2.2588E-05 |
| 3 | Electricity, steam, hot-water | 0.0130 |
| 4 | Gas | 0 |
| 5 | Water | 0.0078 |
| 6 | Coal, coal mining products | 1.5840E-05 |
| 7 | Other mining products | 0 |
| 8 | Crude petroleum, natural gas | 0.0027 |
| 9 | Chemical products, fertile materials | 0.0322 |
| 10 | Mineral oil products | 0.0051 |
| 11 | Synthetic products | 0.4016 |
| 12 | Rubber products | 0.0001 |
| 13 | Stones and clay products, building materials | 6.7764E-05 |
| 14 | Fine ceramics products | 5.0196E-06 |
| 15 | Glass and glass products | 0.0009 |
| 16 | Iron and steel | 2.5098E-06 |
| 17 | Non-ferrous metals, non-ferrous. semifinished metals | 2.5098E-06 |
| 18 | Foundries products | 0.0002 |
| 19 | Drawing shop products, cold rolling mill | 3.0117E-05 |
| 20 | Steel and light metal products, rail vehicle | 5.0196E-06 |
| 21 | Machinery and equipment | 0.0006 |
| 22 | Office equipment, data processing hardware | 1.0039E-05 |
| 23 | Motor vehicles and equipment | 0.0001 |
| 24/25 | Water-craft/ Aircraft and spacecraft | 0 |
| 26 | Electrical products | 0.0008 |
| 27 | Precision engineering and optical products, watches | 4.2666E-05 |
| 28 | Iron sheet metal goods | 0.0147 |
| 29 | Musical instruments, toys, jewellery and sport articles | 0 |
| 30 | Wood and timber ( lumber) | 0.0002 |
| 31 | Wood products | 0 |
| 32 | Wood-pulp, chemical pulp, paper, paperboard | 0.0007 |
| 33 | Paper and paper-goods | 0.0074 |
| 34 | Printing and duplicating | 0.0001 |
| 35 | Leather, leather products and shoes | 0 |
| 36 | Textiles | 0.0055 |
| 37 | Clothing | 5.0196E-06 |
| 38 | Food products | 2.5098E-06 |
| 39 | Beverages | 0 |
| 40 | Tobacco goods | 5.0196E-06 |
| 41 | Structural and civil engineering services | 2.0078E-05 |
| 42 | Finishing trades | 0.0005 |
| 43 | Wholesale services, Recycling | 0.0083 |
| 44 | Retail trade services | 0.0002 |

| No. | Sectors | Input-coefficients |
|---|---|---|
| 45 | Railways services | 0.0008 |
| 46 | Shipping, waterways and harbours services | 8.0313E-05 |
| 47 | Communications | 0.0048 |
| 48 | Other Transport services | 0.0678 |
| 49 | Financial services | 8.5158E-05 |
| 50 | Insurance services | 0.0006 |
| 51 | Building, accomodation services | 0.0026 |
| 52 | Restaurants and hotels services | 0.0008 |
| 53 | Educational, scientific, cultural and sports services | 0.0002 |
| 54 | Medical and veterinary services | 0 |
| 55 | Other business services | 0.0739 |
| 56 | State and local government enterprises | 0.0085 |
| 57/58 | Social insurance enterprises/Private households, non-profit org. | 0 |

**Table 3.15** Age structure of refrigerators in Germany

| | up to 6 years | 6 -10 years | above 10 years | total |
|---|---|---|---|---|
| primary refrigerator | 43% | 34% | 23% | 100% |
| secondary refrigerator | 13% | 40% | 47% | 100% |
| weighted total | 40% | 35% | 26% | 100% |

Source: Baur *et al.* (1994)

**Table 3.16** Refrigerators used in the reference scenario and the progressive scenario

| Technology | refrigerator (class A) with ***-box | refrigerator (class A) without ***-box | refrigerator (class B) with ***-box | refrigerator (class B) without ***-box | adapted refrigerator (A) with ***-box (Scenario I) | adapted refrigerator (A) without ***-box (Scenario II) |
|---|---|---|---|---|---|---|
| Cooling volume (litre) | 114 | 148 | 126 | 162 | 126 | 162 |
| Freezing volume (litre) | 18 | | 18 | | 18 | |
| Additional insulation (litre) | 12 | 14 | 0 | 0 | 13 | 15 |
| Energy consumption (kWh/a) | 161 | 120 | 208 | 201 | 173 | 163 |

the final assembly stage. Indirect inputs are necessary to produce the direct inputs. The burdens are split into pollutants and micropollutants.

**Table 3.17** Burdens of the refrigerator scenarios due to the different *production* steps

| Production level | SCENARIO I | | | | SCENARIO II | | | |
|---|---|---|---|---|---|---|---|---|
| | Direct inputs | Indirect inputs | Final assembly | Total | Direct inputs | Indirect inputs | Final assembly | Total |
| Pollutants(in t) | | | | | | | | |
| $SO_2$ | 5.17 | 9.03 | 1.00 | 15.20 | 6.05 | 10.57 | 1.17 | 17.79 |
| $NO_x$ | 11.98 | 12.86 | 1.13 | 25.97 | 14.02 | 15.05 | 1.32 | 30.40 |
| TSP | 1.09 | 2.19 | 0.08 | 3.36 | 1.28 | 2.56 | 0.09 | 3.93 |
| CO | 5.97 | 10.41 | 1.14 | 17.52 | 6.99 | 12.19 | 1.33 | 20.51 |
| VOC | 2.31 | 2.62 | 0.27 | 5.20 | 2.70 | 3.07 | 0.32 | 6.09 |
| $CO_2$ | 2,509 | 3,591 | 576 | 6,676 | 2,937 | 4,204 | 674 | 7,815 |
| Micropollutants (in kg) | | | | | | | | |
| Pb | 1.485 | 4.327 | 0.307 | 6.119 | 1.738 | 5.065 | 0.359 | 7.163 |
| Cd | 0.037 | 0.132 | 0.005 | 0.174 | 0.043 | 0.155 | 0.006 | 0.204 |
| As | 0.549 | 1.460 | 0.038 | 2.047 | 0.643 | 1.709 | 0.044 | 2.396 |
| Hg | 0.167 | 0.337 | 0.005 | 0.509 | 0.195 | 0.394 | 0.006 | 0.596 |

calculated with EMI2.0

*Avoided burdens calculated with EMI2.0*

Within the ExternE-project, external costs of different technologies should not be compared directly. Nevertheless, we feel that it is necessary to provide orders of magnitude between the burdens due to the production of end-use technologies and the avoided burdens due to using these technologies. Hence, the burdens given in Table 3.17 compare to the avoided burden due to lower electricity needs of the class A refrigerators given in Table 3.18. The values given in that table again are calculated with EMI2.0. The burdens are due to the avoided electricity demand of private households which in Scenario I sums up to *15.7 TWh and 16.9 TWh* in Scenario II. Pollution avoided due to direct inputs is caused e.g. by burning coal for electricity production with the German electricity mix of 1990. Pollution due to indirect inputs is caused e.g. by the German coal mining sector. Burdens due to imported inputs are not considered in the EMI2.0-model.With the electricity price for private households in Germany of $DM_{90}$ 0.2145 per kWh (excluding coal mining subsidies and VAT) the total avoided monetary demands of electricity of $DM_{90}$ 3,366 million (Scenario I) respectively $DM_{90}$ 3,622 million in Scenario II result as input-data for the EMI2.0-calculations. The results reveal a difference of 3 to 4 orders of magnitude between the burdens due to the production of more efficient refrigerators and the avoided burdens due to the reduction of the private households electricity demand in the German economic structure of 1990. With a less polluting energy mix of other countries (e.g. Norway), however, the relationship between additional burdens and avoided burdens of end-use technologies is less dramatic from an electricity production point of view.

**Table 3.18** Avoided burdens due to using class A refrigerators in the different scenarios

| Production level | Scenario I | | | Scenario II | | |
|---|---|---|---|---|---|---|
| | Direct inputs | Indirect inputs | Total | Direct inputs | Indirect inputs | Total |
| Pollutants(in t) | | | | | | |
| $SO_2$ | 24,090 | 2,520 | 26,610 | 25,920 | 2,710 | 28,630 |
| $NO_x$ | 29,210 | 3,430 | 32,640 | 31,430 | 3,690 | 35,120 |
| TSP | 3,330 | 830 | 4,160 | 3,580 | 890 | 4,480 |
| CO | 1,520 | 2,980 | 4,500 | 1,640 | 3,210 | 4,840 |
| VOC | 0,340 | 670 | 1,010 | 370 | 720 | 1,090 |
| $CO_2$ | 10,164,000 | 1,086,000 | 11,250,000 | 10,937,000 | 1,168,000 | 12,105,000 |
| Micropollutants (kg) | | | | | | |
| Pb | 4,153 | 1,054 | 5,207 | 4,469 | 1,134 | 5,603 |
| Cd | 178 | 28 | 206 | 192 | 30 | 222 |
| As | 2,876 | 334 | 3,210 | 3,095 | 359 | 3,454 |
| Hg | 1,088 | 145 | 1233 | 1,171 | 156 | 1,327 |

calculated with EMI2.0

*External cost calculations with EcoSense*

The basic problems of adapting the ExternE-methodology to the evaluation of end-use technologies have already been described in section 3.2.2 (cf. points 2,3 and 5). Many of these problems are due to the site-specific approach. Emissions due to the different industrial activities considered in the EMI2.0-model are dispersed all over Germany. Often, only impacts on the last assembly stage of a production process can be localised and related e.g. to the site of a factory. With respect to the production of insulation material, we have chosen BASF as a typical and large producer. In this case it is reasonable to assume that all upstream processes are located at the same site. The production site of BASF near Mannheim (south-west Germany) is one of the biggest industrial areas in Europe with its own energy supply system. Nevertheless, it is necessary to simplify substantially when using EcoSense, e.g. when assuming that all pollutants are set free in one artificial chimney with a 100m stack height and 4m diameter. During this study the ExternE core-group changed major assumptions of the damage cost assessment. The base year was chagend from 1990 to 1995 and the monetary values adjusted respectively. The recommended exposure-response functions were changed as well. Forest damages have been taken out of the evaluation. The change in evaluating the damage costs of mortality has a dramatic impact on the results. The monetary value of a mortality impact is now based on a concept looking at years of living lost (YOLL) instead of the former concept of value of statistical live (VSL). The evaluation according to YOLL is given in the text whereas the external cost calculations before these changes are given in the annex. Hence, the results of the annex tables are calculated on the basis of VSL, with other exposure-response functions and with a former version of Ecosense (version 1.03 instead of version 2.0). The latter includes an extensive update of the receptor database.

In the older evaluation modus two subtotals for health damages were presented with one excluding chronic mortality impacts. Since the YOLL methodology accounts for the major critisism concerning the evaluation of chronic mortality evaluation the result given in the text do always include these damages. However, there is a distinction in discount rates. The results are presented either assuming a discount rate for human life of 0 percent or 3 percent.

The incremental burdens due to the production of class A refrigerators instead of class B refrigerators were modelled in EcoSense. Table 3.19 gives the results of the EcoSense-run. The upper part of the table gives the results for Scenario I (with ***-compartment) whereas the lower table illustrates the results for refrigerators without ***-compartment (Scenario II). The external costs refer to the amount of electricity that is saved by using class A refrigerators.

**Table 3.19** External costs due to the incremental burdens caused by the production of class A refrigerators instead of class B refrigerators in the different scenarios

| SCENARIO I | Acute mortality (mECU/kWh) | | Chronic mortality (mECU/kWh) | | Morbidity (mECU/kWh) | TOTAL (mECU/kWh) | |
|---|---|---|---|---|---|---|---|
| Health damage | Yoll; 0% | Yoll; 3% | Yoll; 0% | Yoll; 3% | | Yoll; 0% | Yoll; 3% |
| *TSP* | *3.10E-05* | *4.90E-05* | *4.28E-03* | *3.68E-03* | *4.81E-04* | *4.79E-03* | *4.21E-03* |
| *Sulphate* | *9.57E-05* | *1.51E-04* | *1.30E-02* | *1.12E-02* | *1.40E-03* | *1.45E-02* | *1.28E-02* |
| *Nitrate* | *1.85E-04* | *2.93E-04* | *2.56E-02* | *2.20E-02* | *2.84E-03* | *2.86E-02* | *2.51E-02* |
| *$SO_2$* | *2.70E-04* | *4.27E-04* | | | *8.28E-06* | *2.78E-02* | *4.35E-04* |
| Sub-total health | 5.82E-04 | 9.21E-04 | 4.29E-02 | 3.69E-02 | 4.73E-03 | 4.82E-02 | 4.25E-02 |
| Crop damage | | | | | | 8.49E-05 | 8.49E-05 |
| Material damage | | | | | | 9.13E-04 | 9.13E-04 |
| TOTAL | | | | | | 4.92E-02 | 4.35E-02 |

| SCENARIO II | Acute mortality (mECU/kWh) | | Chronic mortality (mECU/kWh) | | Morbidity (mECU/kWh) | TOTAL (mECU/kWh) | |
|---|---|---|---|---|---|---|---|
| Health damage | Yoll; 0% | Yoll; 3% | Yoll; 0% | Yoll; 3% | | Yoll; 0% | Yoll; 3% |
| *TSP* | *3.37E-05* | *5.33E-05* | *4.65E-03* | *4.00E-03* | *5.23E-04* | *5.21E-03* | *4.58E-03* |
| *Sulphate* | *1.04E-04* | *1.65E-04* | *1.42E-02* | *1.22E-02* | *1.52E-03* | *1.58E-02* | *1.39E-02* |
| *Nitrate* | *2.01E-04* | *3.18E-04* | *2.78E-02* | *2.39E-02* | *3.09E-03* | *3.11E-02* | *2.73E-02* |
| *$SO_2$* | *2.93E-04* | *4.63E-04* | | | *9.00E-06* | *3.02E-02* | *4.72E-04* |
| Sub-total health | 6.32E-04 | 1.00E-03 | 4.66E-02 | 4.01E-02 | 5.15E-03 | 5.24E-02 | 4.63E-02 |
| Crop damage | | | | | | 9.23E-05 | 9.23E-05 |
| Material damage | | | | | | 9.92E-04 | 9.92E-04 |
| TOTAL | | | | | | 5.35E-02 | 4.73E-02 |

calculated with EcoSense2.0

**Table 3.20** Global warming damages due to the incremental burdens caused by the production of class A refrigerators instead of class B refrigerators in the different scenarios calculated with EXTERNE results for global warming damage costs per tonne $CO_2$ emitted

| | SCENARIO I | SCENARIO II |
|---|---|---|
| 95% confidence interval | 0.0016–0.058 | 0.0018–0.063 |
| illustrative restricted range | 0.0077–0.019 | 0.0084–0.021 |

*Other impact pathways and external costs*
The calculation do not reflect the impacts of the production of additional amounts of sheet metal and varnish due to bigger refrigerator dimensions of the class A type compared to the class B type when referring to the same cooling and freezing volume. We consider these impacts to be low compared to the impacts due to the production of insulation material.

Further, the calculations above do not reflect the impacts of $CO_2$ on global warming which might be the most severe burden on the medium to long term. In Scenario I, about 6,700 tons of $CO_2$ are emitted to produce refrigerators (7,800 tons in Scenario II). Employing the illustrative restricted range of the ExternE results (see Appendix VII) for global warming damage costs of 0.008–0.02 mECU/kWh are quantified for both scenarios (see Table 3.20). In order to produce the energy that can be saved by using efficient refrigerators in the German electricity system of 1990, about 11 million tons of $CO_2$ would have been emitted in Scenario I and 12 million tons of $CO_2$ in Scenario II. This translates into external costs of 13.2–32.1 mECU/kWh (illustrative restricted range) in both scenarios.

The impact pathway analysis for Scenario II with EcoSense for the micropollutants arsenic and cadmium (total production) provided values for arsenic of mECU 1.302E-06/kWh (lower value) and mECU 1.350E-05/kWh (upper value). The range of external costs due to the local impacts of cadmium emissions are in the range of mECU 5.120E-07/kWh (US/EPA unit risk factor) and mECU 3.414E-06/kWh (LAI risk factor). These pollutants do not have a considerable impact on the external costs of refrigerators because they are by 3 to 4 orders of magnitude lower than the health impacts of nitrates and sulphates. The threshold value for increases in the mercury concentration is not reached due to the emissions of the production process.

Since we only look at incremental burdens of efficient refrigerators compared to less efficient devices, there are not many additional burdens besides those due to the production of additional insulation material. For example, occupational health impacts can only occur when producing additional insulation material. Since we are not looking at the pre-production processes alone, it is reasonable to assume that the occupational health risk of the entire process chain does not differ very much from the average occupational health risk in the German industry. According to the chosen methodology for the assessment of these damages, we assume that the incremental risk of producing insulation material is zero and, thus, no additional damages are evoked due to occupational effects.

In Hohmeyer *et al.* (1996) it is suggested to compare not only the mere impacts of standard technologies and energy efficient devices, but also consider the impacts of spending a given amount of money, i.e. if the amount of money spent for the old technology plus the related

energy consumption differed from the amount of money spent for the energy efficient technology plus the related energy consumption, it would be worthwhile thinking about the impacts of spending the money that can be saved by choosing the cheaper option. Since the monetary value of the energy savings ($DM_{90}$ 3,366 million/3,622 million) by far exceeds the amount of money necessary to increase the energy efficiency of refrigerators ($DM_{90}$ 34 million/40 million), the impacts of spending the monetary savings by buying class A refrigerators could dominate the overall impact, e.g. by buying bigger cars or renting bigger flats. However, this requires that the price difference of efficient refrigerators would only equalise the costs of the additional insulation material. Presently, the private households have to pay much higher prices for efficient refrigerators. Presently a class A refrigerator costs about $DM_{90}$ 70 more than a class B refrigerators. Hence, the additional investment for private households for class A refrigerators amounts to about $DM_{90}$ 2,063 million.Obviously, the manufacturers have partly internalised the monetary savings of the private households in their pricing strategy. This would enable the "innovative" private household who spends his money on class A refrigerators to either save or consume additional $DM_{90}$ 1,303 million/1,586 million. Accordingly, the spending of monetary savings for private households could play a considerable role in the scenarios discussed.

*Conclusions*

Health damages dominate the external costs due to the production of refrigerators with higher energy efficiency. However, with a total of mECU 0.0492 per kWh electricity saved, with 0% discount rate and mECU 0.0435 with 3% for Scenario I (mECU 0.0535and 0.0473 respectively for Scenario II), the impact of this end-use technology seems very low. Even when considering $CO_2$-emissions, the impacts due to the production of energy efficient refrigerators do not exceed this order of magnitude. Although, one has to keep in mind that only their local impacts are accounted for. Within the scope of ExternE, the external costs of the electricity production by modern coal fired power plants were about three orders of magnitude higher than the external costs of energy efficient refrigerators. Including the local impacts of the micropollutants arsenic, cadmium and mercury, according to the exposure-response functions given above, does not have a sizeable impact on the overall results. Considering equal expenditures of private households in energy and refrigerators indicated that the impacts of spending the monetary savings due to buying class A refrigerators could have a considerable impact on the overall results. The changes in exposure-response functions and monetary values during the project has reduced the external costs (Scenario I: mECU 0.124 to 0.492 per kWh; Scenario II: mECU 0.135 to 0.0535 per kWh) by half an order of magnitude.

***Impacts of windows***

The methodology to assess the impacts of triple pane instead of double pane windows is basically the same as the one used to assess the impacts of refrigerators. The incremental inputs that are necessary to switch from a window with two panes to a more energy conserving one with three window panes have already been assessed by Hohmeyer *et al.* (1985) on behalf of the European Commission in 1985. In an interview campaign, eight craftsmen doing the final assembly of windows were asked about the additional inputs (in monetary values) necessary for assembling windows with three panes instead of two. The results of this study have been

**Table 3.21** Input-structure for the production of one triple-glassed instead of one double-glassed window

| No. | Sectors | Incremental inputs: Double -> Triple (one window) in $DM_{90}$ |
|---|---|---|
| 1-8 | | 0.0000 |
| 9 | Chemical products | 1.4803 |
| 10 | Mineral oil products | 0.0000 |
| 11 | Synthetic products | 2.4881 |
| 12 | Rubber products | 1.8036 |
| 13 | Stones and clay products, building materials | 0.0372 |
| 14 | Fine ceramics products | 0.0000 |
| 15 | Glass and Glass products | 12.9642 |
| 16 | Iron and steel | 0.0000 |
| 17 | Non-ferrous metals, non-ferrous. semifinished metals | 1.2608 |
| 18-27 | | 0.0000 |
| 28 | Iron sheet metal goods | 0.3352 |
| 29 | Musical instruments, toys, jewellery and sport articles | 0.0000 |
| 30 | Wood and timber ( lumber) | 0.0000 |
| 31 | Wood products | 1.0492 |
| 32-41 | | 0.0000 |
| 42 | Finishing trades | 10.7191 |
| 43-58 | | 0.0000 |

transferred to our base year 1990 by using price indices of the sectors where the additional inputs are produced (Table 3.21).

*Final demand of windows*

The final demand for windows in Germany depends on the economic situation of the construction business as a whole. The construction sector can be divided into two parts, the construction of new buildings (64% of window demand) and the renovation of already existing buildings (36% of window demand). Given the different juridical circumstances and the diverging statistical data, the evaluation of the building stock in Germany has to be divided into the western and eastern part of Germany. Until 1977, in Western Germany the German Thermal Insulation Ordinance (GTIO) has been the main landmark regarding the heat insulation condition of the building stock; primarily single pane windows with a k-value of max. 5.2 were installed into buildings. The GTIO in 1977 required the installed windows to meet a standard of at least $k < 3.1$ which could be fulfilled by double pane windows.

In the former East Germany mainly double pane windows were installed after 1945. In rooms with low day usage (e.g. sleeping and storage rooms) also single pane windows were installed. As of 1986 double-pane thermal insulation windows became the standard. Since the German unification the GTIO (currently the 1995 update) also applies to the eastern part of Germany.

**Table 3.22** Window stock in the residential buildings built in West Germany before 1990.

| built in | k-value | living space [mill. $m^2$] | window surface [mill. $m^2$]* |
|---|---|---|---|
| until 1977 | < 5.2 (single pane) | 1918 | 364.4 |
| 1978 - 1990 | < 3.1 (double pane) | 339 | 64.4 |

* mean ratio window surface/living space = 0.19

**Table 3.23** Window-stock in the residential buildings built in East-Germany until 1990.

| built in | k-value | living space [mill. $m^2$] | window surface [mill. $m^2$]* |
|---|---|---|---|
| until 1945 | < 5.2 (single pane) | | |
| 1946 - 1985 | < 3.5 (double pane) | 394 | 74.9 |
| 1986 - 1990 | < 2 (thermal insulation) | 25.9 | 4.9 |

* mean ratio window surface/living space = 0.19

**Table 3.24** Market shares of different frame materials in Germany between 1994 - 1996

| | 1994 | | 1995* | | 1996* | |
|---|---|---|---|---|---|---|
| | windows [mill.] | market share | windows [mill.] | market share | windows [mill.] | market share |
| Total | 25.164 | 100 % | 26.635 | 100 % | 25.960 | 100 % |
| Wood | 7.556 | 30.0 % | 7.738 | 29.0 % | 7.300 | 28.1 % |
| Plastic | 11.773 | 46.8 % | 12.670 | 47.5 % | 12.300 | 47.4 % |
| Aluminium | 5.235 | 20.8 % | 5.529 | 20.8 % | 5.517 | 21.3 % |
| Aluminium/ Wood | 0.600 | 2.4 % | 0.698 | 2.6 % | 0.844 | 3.2 % |

* values of 1995 and 1996 are estimates

Source: Window Trade Association of Germany

In 1995 the market volume of the window-market in Germany was 33 million $m^2$. The total demand for windows is split into four different kinds of frame materials (Table 3.24 gives an overview). The highest share of almost 50% is held by plastic, namely poly-vinyl-chloride (PVC) compounds. The remaining is split between wood (~30%) and aluminium (~20%). The market share of the wood-aluminium compound (~3%) can be neglected.

*Scenario*

In order to calculate the external costs of technologically superior windows compared to standard thermal insulation windows the following scenario is assumed: The window market will be regarded over the average lifetime of a window. Several authors assume the lifetime of windows to be between 25 and 50 years. A conservative estimate can certainly be set to 25 years. Regarding the mean annual market volume of 25.92 mill. windows, two alternative developments over the next 25 years will be compared:

The reference path is based on the market situation of 1990 which will be extrapolated until 2015. This means that every year about 26 million windows will be installed, either in newly constructed or renovated buildings which just fulfill the German Thermal Insulation Ordinance (double pane windows). The alternative path assumes that the same amount of windows will be installed every year until 2015, but with triple pane windows. The emissions due to the production and installation and the emissions originating from the heat loss of the building because of the differing insulation properties of the windows, will be calculated separately. Even though we are looking at a time-period of 25 years, we assume that all window installations and all energy savings will take place in 1990. Thus, the total amount of windows under consideration is 25 times the amount of annual window installations which is assumed to be about 26 million. Hence, a total demand for about 650 million windows (with a surface area of about 825 million $m^2$) will determine the input-vector for the EMI2.0-calculations.

The energy-loss of a building caused by a window is given by the heat flow through the surface of the window-opening in the wall. The energy-savings due to the installation of a better insulating window are caused by a smaller heat flow through the surface. The heat flow is parametrisised by the k-value of the window. Therefore the potential energy-savings $E_{savings}$ per year is a function of the difference of the k-value $k_b$ of the old window which is replaced by a new one with $k_a$: $k_d = k_b - k_a$

The potential energy-savings $E_{savings}\ (k_d)$ per year is given by the following formula (Feist, 1986):

$$E_{savings}\ (k_d) = A \cdot (k_b - k_a) \cdot (T_{in} - T_{out}) \cdot 6{,}552\ h/a \cdot f_B$$

| | |
|---|---|
| $E_{savings}$ | energy-savings |
| $A$ | surface of window |
| $k_b$ | k-value before (two layers: 1.4) |
| $k_a$ | k-value after (three layers: 0.78) |
| $T_{in}$ | temperature on the inside of window (mean: 290.15 K) |
| $T_{out}$ | outdoor temperature (mean: 279.49 K) |
| $f_B$ | factor correcting behaviour of building (cool down at night (max. 0.95)) and passive solar energy-input through window surface (max. 0.96) |

$A$ is the surface of the window opening in the wall. Besides the transparent surface this includes the frame as well as the roller blind box. $T_{in} - T_{out}$ represents the average temperature difference between the outside and the inside surface of the window. The heat-loss is only accountable during the heating-period which in Germany lasts from September until May. This adds up to an average of 6,552 hours per year. From meteorological data the average outdoor temperature $T_{out}$ during the heating period in Germany is 6.34 °C (279.49 K). The indoor temperature $T_{in}$ at the surface of the window pane is considered to be 17 °C the minimum. It has to be taken into account that $T_{in}$ does not represent the average room-temperature but the temperature at the surface of the window. The room temperature at the window-surface is in principle lower than the temperature in the centre of the room during the heating period. $f_B$ is a factor correcting for the interaction of the building. It is itself split into a factor considering

**Table 3.25** Energy-mix used for heating of private households in Germany

| Type | Portion* | Price [$DM_{90}$/kWh] § |
|---|---|---|
| Oil | 47.8 % | 0.03 |
| Gas | 35.1 % | 0.05 |
| Coal | 4.2 % | 0.07 |
| Electricity | 5.7 % | 0.10 (night rate) |
| District heating | 5.0 % | 0.06 |
| Other | 2.2 % | |

Source: *Enquete (1995); §Hohmeyer *et al.* (1996)

the cooling down of the whole building at night and one allowing for the passive solar energy input caused by the radiation input.

Putting in the actual values for the parameters described above results in an annual energy-savings $E_{annual\ savings}$ of:

$$E_{annual\ savings} = 825{,}000{,}000\ m^2\ (1.4\text{-}0.78)W/(m^2K)(290.15\text{-}279.49)K\ 6{,}552\ h/a\ 0.912$$
$$= 32.6\ TWh/a$$

Hence, the total energy saving in the 25 year period is about 814.54 TWh end use energy. In order to be able to calculate the avoided burden due to lower heat-losses in buildings we consider the average energy-mix used for heating buildings. Table 3.25 lists the statistical energy-mix used for heating of private households in Germany.

The result of the transformation of the energy-savings into an input vector for EMI2.0 is shown in Table 3.26.

*Burdens of triple pane windows calculated with EMI2.0*

Using the incremental input-structure of a three layer window compared with a two layer window and the number of windows in the scenario, the input-data for calculations with EMI2.0 can be generated by multiplying the input-vector with the total amount of windows. The burdens calculated with EMI2.0 are illustrated in Table 3.27. The direct inputs reflect the production process of the material that a window-craftsmen uses for the final assembly of the window (including glass production).

Local impacts can only be related to the site where the windows are assembled. Nevertheless, it is interesting to calculate the external costs due to all upstream processes and the final assembly as well. Consequently, we have conducted two EcoSense-runs to assess the external costs of window production.

**Table 3.26** Annually avoided demand for end energy sources due to the installation of about 650 million windows

| No. | Sector | Demand (in million $DM_{90}$) |
|---|---|---|
| 3 | Electricity, steam, hot water* | 7,226.688 |
| 4 | Gas | 12,920.196 |
| 6 | Coal | 2,464.561 |
| 10 | Mineral oil products | 10,100.181 |

* includes district heating

**Table 3.27** Burdens of window production and pre-production processes

| Production level | Direct inputs | Indirect inputs | Final assembly | Total |
|---|---|---|---|---|
| Pollutants (in t) | | | | |
| $SO_2$ | 15,700 | 10,110 | 130 | 25,940 |
| $NO_x$ | 19,180 | 14,830 | 1,330 | 35,340 |
| TSP | 2,510 | 2,500 | 70 | 5,080 |
| CO | 8,320 | 12,980 | 4,200 | 25,500 |
| VOC | 2,050 | 2,510 | 840 | 5,400 |
| $CO_2$ | 2,952,970 | 3,916,600 | 147,870 | 7,017,440 |
| Micropollutants (in kg) | | | | |
| Pb | 12,454 | 6,472 | 11 | 18,937 |
| Cd | 2,029 | 435 | 1 | 2,465 |
| As | 32,566 | 5,876 | 5 | 38,447 |
| Hg | 22 | 299 | 0 | 321 |

calculated with EMI2.0

*Avoided burdens calculated with EMI2.0*

This study focuses on the external cost accounting of end-use technologies. Avoided external costs due to energy savings are not taken into account. However, in order to get a feeling for the orders of magnitude of the burdens caused by producing end-use technologies and those avoided due to energy savings, we present the emissions that are avoided due to using efficient windows in the given scenario. It must be stressed that the results heavily depend on the energy mix in Germany of 1990 and will be different in countries with differenet energy mixes.

The assessment of avoided emissions due to the reduced demand for energy inputs in private households is based on the energy input structure for space heating given above. The avoided emissions have partly been calculated with EMI2.0. Emissions due to burning energy inputs in private households (e.g. natural gas, coal, light oil) have been calculated with average emission coefficients of the year 1988). The results are given in Table 3.28. Direct inputs in the table, e.g., represent the emissions of a truck delivering light oil to private households. Emissions due to indirect inputs are caused, e.g. by the production of the gasoline for the truck that delivers lightoil to private households. Emissions due to final burning occur when light oil,

**Table 3.28** Avoided burdens due to energy savings

| Energy production level | Direct inputs† | Indirect inputs† | Final burning | Total |
|---|---|---|---|---|
| Pollutants (in t) | | | | |
| $SO_2$ | 69,157 | 10,799 | 181,235 | 261,191 |
| $NO_x$ | 69,214 | 15,945 | 127,703 | 212,862 |
| TSP | 8,802 | 3,544 | 32,995 | 45,342 |
| CO | 9,351 | 15,478 | 740,621 | 765,449 |
| VOC | 7,709 | 3,426 | 49,102 | 60,237 |
| $CO_2$ | 24,677,859 | 5,058,000 | 172,814,428 | 202,550,279 |
| Micropollutants (in kg) | | | | |
| Pb | 9,575 | 5,450 | 11,959 | 26,984 |
| Cd | 400 | 125 | 4,623 | 5,148 |
| As | 6,400 | 1,375 | 12,685 | 20,460 |
| Hg | 2,575 | 550 | 1,148 | 4,273 |

† calculated with EMI2.0

coal or natural gas is burned to heat private homes. Obviously, the emissions avoided due to energy savings are an order of magnitude higher than the emissions due to the incremental production effects of the high-efficiency windows.

*External costs of windows calculated with EcoSense*
In order to have a realistic scenario, we have chosen Gelsenkirchen as the site where all the emissions from window production processes are generated. Since the majority of the emissions occur in the glass production process, and one of the large glass manufacturing companies is located in Gelsenkirchen, we thought it was an appropriate site to choose for the scenario.

The results of the EcoSense runs for Gelsenkirchen and the window-scenario are given in 3.29.

*Other impact pathways*
Looking at the $CO_2$-emissions reveals additional external costs due to the contribution to global climate change. Employing the illustrative restricted range of the ExternE results (see Appendix VII) for global warming damage costs of 0.003–0.008 mECU/kWh energy saved are quantified for the assembly of windows and 0.2–0.4 mECU/kWh for the total production (see Table 3.30). This compares to a reduction of $CO_2$-emissions due to avoided energy production with an global warming damage cost potential of 4.6–11.2 mECU/kWh energy production.

The local impact assessment for the micropollutants arsenic and cadmium by the exposure-response functions given above resulted in external costs due to the total window production (including all upstream processes) of mECU 2.96E-06 to 5.96E-05 per kWh energy saved for arsenic and mECU 1.69E-06 to 1.13E-05 for cadmium. These values are about one or two

**3.29** Damages caused by the assembly of windows (upper table) and damages of the total process chain (lower table)

| CRAFTSMEN | Acute mortality (mECU/kWh) | | Chronic mortality (mECU/kWh) | | Morbidity (mECU/kWh) | TOTAL (mECU/kWh) | |
|---|---|---|---|---|---|---|---|
| Health damage | Yoll; 0% | Yoll; 3% | Yoll; 0% | Yoll; 3% | | Yoll; 0% | Yoll; 3% |
| *TSP* | *3.43E-05* | *5.43E-05* | *4.74E-03* | *4.08E-03* | *5.27E-04* | *5.30E-03* | *4.66E-03* |
| *Sulphate* | *5.89E-06* | *9.32E-06* | *8.01E-04* | *6.89E-04* | *8.63E-05* | *8.94E-04* | *7.85E-04* |
| *Nitrate* | *1.07E-04* | *1.69E-04* | *1.48E-02* | *1.27E-02* | *1.64E-03* | *1.65E-02* | *1.45E-02* |
| *$SO_2$* | *7.57E-05* | *1.20E-04* | | | *2.32E-06* | *7.80E-05* | *1.22E-04* |
| Sub-total health | 2.23E-04 | 3.53E-04 | 2.03E-02 | 1.75E-02 | 2.26E-03 | 2.28E-02 | 2.01E-02 |
| Crop damage | | | | | | 2.23E-05 | 2.23E-05 |
| Material damage | | | | | | 2.67E-04 | 2.67E-04 |
| TOTAL | | | | | | 2.31E-02 | 2.04E-02 |

| TOTAL PRODUCTION | Acute mortality (mECU/kWh) | | Chronic mortality (mECU/kWh) | | Morbidity (mECU/kWh) | TOTAL (mECU/kWh) | |
|---|---|---|---|---|---|---|---|
| Health damage | Yoll; 0% | Yoll; 3% | Yoll; 0% | Yoll; 3% | | Yoll; 0% | Yoll; 3% |
| *TSP* | *7.27E-04* | *1.15E-03* | *1.00E-01* | *8.61E-02* | *1.13E-02* | *1.12E-01* | *9.86E-02* |
| *Sulphate* | *2.05E-03* | *3.24E-03* | *2.78E-01* | *2.39E-01* | *3.00E-02* | *3.10E-01* | *2.72E-01* |
| *Nitrate* | *2.51E-03* | *3.97E-03* | *3.47E-01* | *2.99E-01* | *3.86E-02* | *3.88E-01* | *3.42E-01* |
| *$SO_2$* | *6.32E-03* | *1.00E-02* | | | *1.94E-04* | *6.51E-03* | *1.02E-02* |
| Sub-total health | 1.16E-02 | 1.83E-02 | 7.26E-01 | 6.25E-01 | 8.00E-02 | 8.18E-01 | 7.23E-01 |
| Crop damage | | | | | | 3.14E-03 | 3.14E-03 |
| Material damage | | | | | | 1.81E-02 | 1.81E-02 |
| TOTAL | | | | | | 8.39E-01 | 7.44E-01 |

calculated with EcoSense2.0

**Table 3.30** Global warming damages caused by the assembly of windows and damages of the total process chain calculated with EXTERNE results for global warming damage costs per tonne $CO_2$ emitted

| | CRAFTSMEN | TOTAL PRODUCTION |
|---|---|---|
| 95% confidence interval | 0.0007–0.04 | 0.003–1.2 |
| illustrative restricted range | 0.003–0.008 | 0.2–0.4 |

orders of magnitude lower than the external costs due to impacts on forests. Accordingly, they literally do not influence the external costs due to the production of efficient windows. The threshold value for mercury given above has not been reached.

Since we only assess incremental damages, the basic source of external costs is the process chain of producing e.g. an additional window pane, additional sealing material, additional

window frame material. We assume that the occupational health risk arising from the entire production process chain is similar to the average occupational health risk in the German industry. Thus , the incremental risk is zero and no external costs due to occupational health risk are accounted for according to the methodology agreed upon in the ExternE-kernel group. The disposal of additional material for triple pane windows can cause damages. However, we are not able to give more detailed damage calculations due to the complexity of the impact pathway and the lack of suitable damage-functions. The aspect of total spending does not play a major role in this scenario since the additional costs of producing a window with three instead of two layers, more or less equals the monetary value of the predicted energy savings.

*Conclusions*

The external costs of increasing the insulation qualities of windows add up to mECU 0.839 per kWh end-use energy saved with 0 percent discount rate. Assuming a 3 percent discount rate results in external costs of mECU 0.744 per kWh end-use energy saved. This is about half an order of magnitude lower than the external costs of a fossil fuel based energy supply technology. The health effects dominate the results. Excluding chronic mortality effects reduces external costs by nearly an order of magnitude. Employing the illustrative restricted range of the EXTERNE results for global warming, would add damage costs of 0.2–0.4 mECU/kWh end-use energy saved for the total production to the results given above. The local impacts due to the micropollutants arsenic and cadmium do not have a considerable influence on the overall results.

***Impacts of heat exchanger***

The heat exchanger scenario is closely linked to a scenario conducted by the German Institute of Economic Research (DIW) on employment effects of energy efficiency measures. The input-structure in the base year 1990, required by the EMI2.0-model can be drawn from this study. The structure is split into inputs for the production process of heat exchangers, inputs necessary to expand the production capacity, and inputs necessary to maintain and modernise the existing production capacity. The respective input vectors are illustrated in Table 3.31.

*Scenario*

The DIW-Scenario considers a 10 year time period (1990 until 2000) with additional (with respect to a reference case) investment in heat exchangers as illustrated in Table 3.32. Over the 10 year period, the total investment in heat exchangers in this scenario is $DM_{90}$ 6.7 billion. In order to be able to calculate comparable results as for the technologies described above, we assume that the total investment will take place in the reference year 1990. This total demand is multiplied with the input-vector of the heat exchanger production. Furthermore, it is assumed that each million DM of additional production capacity requires an investment of 0.67 million DM for capacity expansion. DM 955 million is the peak in additional heat exchanger production. Hence, an investment of about $DM_{90}$ 640 million is required for capacity expansion. This value will be transferred into an EMI2.0-input according to the input-coefficients

**Table 3.31** Input-coefficients of heat exchanger production, the expansion of the production capacity, and of maintaining and modernising the existing production capacity

| No. | Sectors | Input structure | Capacity expansion | Maintenance / Modernisation |
|---|---|---|---|---|
| 1 | Agricultural products | 0 | 0.000091 | 0.000003 |
| 2 | Forestry, fishery products | 0 | 0 | 0 |
| 3 | Electricity, steam, hot-water | 0.004183 | 0 | 0 |
| 4 | Gas | 0.000755 | 0 | 0 |
| 5 | Water | 0.000341 | 0 | 0 |
| 6-9 | | 0 | 0 | 0 |
| 10 | Mineral oil products | 0.012898 | 0 | 0 |
| 11 | Synthetic products | 0 | 0.001169 | 0.00004 |
| 12 | Rubber products | 0.000412 | 0 | 0 |
| 13 | Stones and clay products, building materials | 0.005902 | 0 | 0 |
| 14 | Fine ceramics products | 0 | 0 | 0 |
| 15 | Glass and Glass products | 0.002363 | 0 | 0 |
| 16 | Iron and steel | 0 | 0 | 0 |
| 17 | Non-ferrous metals, non-ferrous. semifinished metals | 0 | 0.000091 | 0.000003 |
| 18 | Foundries products | 0 | 0.011461 | 0.000391 |
| 19 | Drawing shop products, cold rolling mill | 0 | 0.005318 | 0.000181 |
| 20 | Steel and light metal products, rail vehicle | 0.514255 | 0.018404 | 0.000628 |
| 21 | Machinery and equipment | 0.023171 | 0.282225 | 0.009625 |
| 22 | Office equipment, data processing hardware | 0 | 0.046142 | 0.001574 |
| 23 | Motor vehicles and equipment | 0 | 0.032894 | 0.001122 |
| 24 | Water-craft | 0 | 0 | 0 |
| 25 | Aircraft and spacecraft | 0 | 0.001444 | 0.000049 |
| 26 | Electrical products | 0.047344 | 0.085954 | 0.002931 |
| 27 | Precision engineering and optical products, watches | 0 | 0.000413 | 0.000014 |
| 28 | Iron sheet metal goods | 0.086072 | 0.019965 | 0.000681 |
| 29-30 | | 0 | 0 | 0 |
| 31 | Wood products | 0 | 0.022808 | 0.000778 |
| 32 | Wood-pulp, chemical pulp, paper, paperboard | 0 | 0 | 0 |
| 33 | Paper and paper-goods | 0.001105 | 0 | 0 |
| 34 | Printing and duplicating | 0.000755 | 0 | 0 |
| 35 | Leather, leather products and shoes | 0 | 0 | 0 |
| 36 | Textiles | 0 | 0.001307 | 0.000045 |
| 37-40 | | 0 | 0 | 0 |
| 41 | Structural and civil engineering services | 0 | 0.131498 | 0.004485 |
| 42-43 | | 0 | 0 | 0 |

| No. | Sectors | Input structure | Capacity expansion | Maintenance / Modernisation |
|---|---|---|---|---|
| 44 | Retail trade services | 0.000815 | 0 | 0 |
| 45-46 | | 0 | 0 | 0 |
| 47 | Communications | 0.006294 | 0 | 0 |
| 48 | Other Transport services | 0.010096 | 0 | 0 |
| 49 | Financial services | 0.000374 | 0 | 0 |
| 50 | Insurance services | 0.00135 | 0 | 0 |
| 51 | Building, accomodation services | 0.001604 | 0 | 0 |
| 52-54 | | 0 | 0 | 0 |
| 55 | Other business services | 0.040974 | 0.010956 | 0.000374 |
| 56-58 | | 0 | 0 | 0 |

Source: (Umweltbundesamt, 1993)

**Table 3.32** The additional investment in heat exchangers compared to a reference case in the DIW-scenario split up among 7 industrial sectors

| Year<br>Sector | 1992 (million $DM_{90}$) | 1995 (million $DM_{90}$) | 1998 (million $DM_{90}$) | 2000 (million $DM_{90}$) |
|---|---|---|---|---|
| Chemical products | 62 | 86 | 85 | 80 |
| Stones and Clay Products | 118 | 143 | 121 | 107 |
| Iron and steel | 285 | 357 | 168 | 43 |
| Foundries products | 26 | 31 | 28 | 3 |
| Motor vehicles | 31 | 46 | 47 | 36 |
| Electrical products | 18 | 43 | 36 | 36 |
| Food products | 159 | 249 | 127 | 48 |
| Total | 699 | 955 | 612 | 353 |

Source: (Umweltbundesamt, 1993)

for capacity expansion. Maintenance & modernisation requires 2.2 % of the annual turnover, about $DM_{90}$ 147 million. This value completes the necessary input data for EMI2.0-calculations.

According to the table above the total energy savings add up to *1,491 TWh end-use energy* in the ten year time period.

**Table 3.33** End-use energy savings due to the additional investment in heat exchangers according to the DIW-scenario

| Sector<br>Energysource<br>(GWh) | Chem-<br>istry | Stones<br>and clay | Iron and<br>steel | Foundries<br>products | Motor<br>vehicles | Electrical<br>products | Food<br>products |
|---|---|---|---|---|---|---|---|
| Hard coal | 8,129 | 100,235 | 25,921 | 902 | 1,764 | 435 | 11,401 |
| Hard coal coke | 956 | 7,627 | 310,450 | 15,396 | 0 | 261 | 2,700 |
| Crude petroleum | 68,221 | 60,740 | 27,127 | 6,767 | 11,266 | 16,127 | 102,312 |
| Natural gas | 36,979 | 69,456 | 176,022 | 16,749 | 20,938 | 7,129 | 118,514 |
| Industrial electricity | 45,109 | 34,320 | 63,296 | 16,524 | 22,929 | 19,518 | 65,408 |
| Total | 159,394 | 272,377 | 602,815 | 56,339 | 56,897 | 43,469 | 300,335 |

Source: Umweltbundesamt, 1993

*Burdens of heat exchanger production calculated with EMI2.0*
In contrast to refrigerators and windows, the burdens of the heat exchanger production are no incremental burdens, since we assume that we introduce a new technology without replacing a conventional device. One of the big manufacturers of heat exchangers in Germany is located in Bochum in North-Rhine-Westfalia,and we choose this site for our scenario. We conducted two scenario runs. One is only reflecting the burdens due to the last assembly process in the factory in Bochum (Final assembly). The second is assuming that all pre-production and upstream processes as well are located in Bochum (Total production). The emission results are given in Table 3.34 split up into the pollution sources final assembly process, upstream and pre-production processes, expansion of production, and maintenance and modernisation of existing production capacities.

*Avoided burdens due to heat exchanger operation*
The emissions due to the production of heat exchangers have to be compared to the avoided emissions due to energy savings when heat exchangers are in operation. Table 3.35 presents the avoided emissions of the DIW heat exchanger scenario. The emissions due to direct and indirect inputs have been calculated with EMI2.0 whereas the emissions due to final burning have been calculated by using average emission-coefficients of the manufacturing sector for converting fossil primary energy inputs into useful energy. Emissions of direct inputs for instance are due to the refining process of crude oil whereas emissions of indirect inputs reflect for example the burdens due to producing the food for the workers in the refinery. Emissions of final burning represent those burdens that occur when, e.g., crude oil is burned in industrial processes.

The avoided emissions due to energy savings from using the produced heat exchangers are about two orders of magnitude higher than the emissions caused by the production process and production capacity enlargement of heat exchangers. Note that the results confer to the German electricity mix of 1990 and can be different in countries with a considerably different energy mix.

**Table 3.34** Burdens of the DIW heat exchanger scenario

| Pollution sources<br>Pollutants (t) | Final assembly | Upstream & pre-production | Expansion | Maintenance/ modernisation | Total Scenario |
|---|---|---|---|---|---|
| $SO_2$ | 120 | 1,980 | 120 | 1 | 2,221 |
| $NO_x$ | 540 | 3,540 | 270 | 2 | 4,352 |
| TSP | 30 | 1,290 | 80 | 1 | 1,401 |
| CO | 1,001 | 9,450 | 610 | 5 | 11,066 |
| VOC | 250 | 870 | 70 | 1 | 1,191 |
| $CO_2$ | 132,060 | 1,082,120 | 75,340 | 592 | 1,290,112 |
| Micropollutants (kg) | | | | | |
| Pb | 317 | 4,136 | 239 | 2 | 4,694 |
| Cd | 1 | 89 | 4 | 0 | 94 |
| As | 1 | 598 | 24 | 0 | 623 |
| Hg | 0 | 54 | 4 | 0 | 58 |

calculated with EMI2.0

**Table 3.35** Avoided emissions by energy saving in the DIW heat exchanger scenario.

| Pollution sources<br>Pollutants (t) | Direct inputs | Indirect inputs | Final burning | Total Scenario |
|---|---|---|---|---|
| $SO_2$ | 273,500 | 33,550 | 1,031,000 | 1,338,000 |
| $NO_x$ | 338,800 | 48,420 | 343,800 | 731,100 |
| TSP | 38,470 | 11,500 | 93,790 | 143,800 |
| CO | 21,280 | 46,910 | 117,800 | 186,000 |
| VOC | 5,380 | 9,580 | 23,570 | 38,530 |
| $CO_2$ | 119,800,000 | 14,920,000 | 178,900,000 | 313,600,000 |
| Micropollutants (kg) | | | | |
| Pb | 47,370 | 17,040 | 75,364 | 139,800 |
| Cd | 2,015 | 409 | 7,285 | 9,709 |
| As | 32,650 | 4,591 | 97,490 | 134,700 |
| Hg | 12,430 | 1,864 | 12,110 | 26,400 |

† calculated with EMI2.0

*External costs of heat exchangers calculated with EcoSense*

Similar to the external cost evaluation for energy efficient windows, we conducted two EcoSense runs to assess the external costs of heat exchanger production. The first scenario looked only at the final assembly of heat exchangers, while the second run considered all production and pre-production processes. Table 3.36 illustrates the results of the EcoSense runs.

**Table 3.36** External costs due to the final assembly process of heat exchangers in the DIW scenario (upper table) and due to all production and pre-production processes of heat exchanger, production capacity expansion, and maintenance/modernisation (lower table)

| FINAL ASSEMBLY | Acute mortality (mECU/kWh) | | Chronic mortality (mECU/kWh) | | Morbidity (mECU/kWh) | TOTAL (mECU/kWh) | |
|---|---|---|---|---|---|---|---|
| Health damage | (Yoll; 0%) | (Yoll; 3%) | (Yoll; 0%) | (Yoll; 3%) | | (Yoll; 0%) | (Yoll; 3%) |
| *TSP* | *5.25E-06* | *8.30E-06* | *7.26E-04* | *6.25E-04* | *8.10E-05* | *8.12E-04* | *7.14E-04* |
| *Sulphate* | *5.68E-06* | *8.98E-06* | *7.73E-04* | *6.65E-04* | *8.32E-05* | *8.62E-04* | *7.57E-04* |
| *Nitrate* | *3.11E-05* | *4.92E-05* | *4.29E-03* | *3.69E-03* | *4.76E-04* | *4.80E-03* | *4.22E-03* |
| *$SO_2$* | *2.45E-05* | *3.88E-05* | | | *7.52E-07* | *2.53E-05* | *3.96E-05* |
| Sub-total health | 6.65E-05 | 1.05E-04 | 5.79E-03 | 4.98E-03 | 6.41E-04 | 6.50E-03 | 5.73E-03 |
| Crop damage | | | | | | 1.08E-05 | 1.08E-05 |
| Material damage | | | | | | 9.21E-05 | 9.21E-05 |
| TOTAL | | | | | | 6.60E-03 | 5.83E-03 |

| TOTAL PRODUCTION | Acute mortality (mECU/kWh) | | Chronic mortality (mECU/kWh) | | Morbidity (mECU/kWh) | TOTAL (mECU/kWh) | |
|---|---|---|---|---|---|---|---|
| Health damage | (Yoll; 0%) | (Yoll; 3%) | (Yoll; 0%) | (Yoll; 3%) | | (Yoll; 0%) | (Yoll; 3%) |
| *TSP* | *1.60E-04* | *2.53E-04* | *2.22E-02* | *1.91E-02* | *2.47E-03* | *2.48E-02* | *2.18E-02* |
| *Sulphate* | *1.18E-04* | *1.87E-04* | *1.61E-02* | *1.39E-02* | *1.73E-03* | *1.79E-02* | *1.58E-02* |
| *Nitrate* | *2.10E-04* | *3.32E-04* | *2.91E-02* | *2.50E-02* | *3.23E-03* | *3.25E-02* | *2.86E-02* |
| *$SO_2$* | *4.50E-04* | *7.12E-04* | | | *1.38E-05* | *4.64E-04* | *7.26E-04* |
| Sub-total health | 9.39E-04 | 1.49E-03 | 6.73E-02 | 5.79E-02 | 7.44E-03 | 7.57E-02 | 6.69E-02 |
| Crop damage | | | | | | 1.94E-04 | 1.94E-04 |
| Material damage | | | | | | 1.15E-03 | 1.15E-03 |
| TOTAL | | | | | | 7.68E-02 | 6.83E-02 |

calculated with EcoSense2.0

*Other impact pathways*

Including the global warming damage costs would result in additional 0.02–0.04 mECU/kWh end-use energy saved (illustrative restricted range) or in additional 0.003–0.1 mECU/kWh (95% confidence interval). This compares to avoided external costs of $CO_2$-emissions due to avoided energy production of 3.7–9.5 mECU/kWh or 0.8–28.5 mECU/kWh supplied energy, respectively.

The impact analyses for the micropollutants arsenic (mECU 1.09E-05 per kWh energy saved to mECU 2.18E-04/kWh) and cadmium (mECU 9.80E-05/kWh to mECU 6.53E-04/kWh) do not provide considerable values. The mercury concentration in the air in the neighbourhood of the heatexchanger factory does not reach the threshold value of 30 µg/person.

Since we look at the entire process chain of the production of heat exchangers, it seems to be reasonable to assume that there is no difference between the average occupational health risk of the entire process chain and the German industry average. Hence, no external costs of occupational health impacts are accounted for. Since the use of heat exchangers is considered to be economically viable, the impacts of spending the monetary savings can be important for the results of external cost assessment. If the president of a steel company privatised all the monetary savings of using heat exchangers, and spent all the money on holiday flights to places far away from his home, the external costs of spending this monetary savings would definitely be higher than those given above. This impact is rather speculative and, thus, difficult to quantify. But it indicates that realising economic energy efficiency measures does not reduce emissions as predicted by solely looking at the impacts of the chosen technology.

*Conclusion*
Even when considering the impacts of the total burdens due to the heat exchanger production, capacity extension and modernisation & maintenance according to the DIW-scenario, the external costs related to the achievable energy savings are extremely low. They sum up to mECU 0.077 per kWh end energy with a 0 percent discount rate (mECU 0.068/kWh assuming a 3 percent discount rate). This is about half an order of magnitude lower than the results in the former evaluation mode. Looking at the final assembly process leads to results which are approximately one order of magnitude lower. Again, health impacts dominate the external cost accounting. Including external costs of $CO_2$-emissions would not change these results dramatically. Employing the EXTERNE global warming damage costs of 0.02–0.04 mECU/kWh end-use energy saved (illustrative restricted range) are quantified. The micropollutants arsenic, cadmium and mercury do not have a considerable local impact on the overall results when assessing them with the exposure-response functions introduced above.

Nevertheless, heat exchangers are in many cases not only an economically viable but an environmentally sound solution as well for responding to growing energy needs. The external costs of their production are about two to three orders of magnitude lower than the external costs of fossil based energy supply systems.

### 3.2.4 References

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# ANNEX

The results in this annex are calculated on the same basis as former ExternE-results, i.e.on the basis of VSL, with other exposure-response functions and with an older version of Ecosense (version 1.03 instead of version 2.0 used above). The latter includes an extensive update of the receptor database. The monetary values are calculated with factor prices of the year 1990 instead of the 1995 prices used above.

**Former Table 3.19** External costs due to the incremental burdens caused by the production of class A refrigerators instead of class B refrigerators in the different scenarios under the old evaluation modus.

| SCENARIO I | Acute mortality (mECU/kWh) | Chronic mortality (mECU/kWh) | Morbidity (mECU/kWh) | Sub-totals incl. chronic mortality Scenario I (mECU/kWh) | Sub-totals excl. chronic mortality Scenario I (mECU/kWh) |
|---|---|---|---|---|---|
| Health damage | | | | | |
| TSP | 3.11E-03 | 1.15E-02 | 6.66E-04 | 1.53E-02 | 3.78E-03 |
| Sulphate | 4.90E-03 | 1.82E-02 | 1.05E-03 | 2.42E-02 | 5.95E-03 |
| Nitrate | 1.72E-02 | 6.38E-02 | 3.68E-03 | 8.46E-02 | 2.09E-02 |
| Crop damage | | | | 0.00E+00 | 0.00E+00 |
| Material damage | | | | 3.45E-04 | 3.45E-04 |
| Forest damage | | | | 2.71E-05 | 2.71E-05 |
| TOTAL | | | | 1.24E-01 | 3.10E-02 |

| SCENARIO II | Acute mortality (mECU/kWh) | Chronic mortality (mECU/kWh) | Morbidity (mECU/kWh) | Sub-totals incl. chronic mortality Scenario I (mECU/kWh) | Sub-totals excl. chronic mortality Scenario I (mECU/kWh) |
|---|---|---|---|---|---|
| Health damage | | | | | |
| TSP | 3.38E-03 | 1.26E-02 | 7.24E-04 | 1.67E-02 | 4.11E-03 |
| Sulphate | 5.33E-03 | 1.98E-02 | 1.14E-03 | 2.63E-02 | 6.48E-03 |
| Nitrate | 1.87E-02 | 6.94E-02 | 4.00E-03 | 9.20E-02 | 2.27E-02 |
| Crop damage | | | | 0.00E+00 | 0.00E+00 |
| Material damage | | | | 3.75E-04 | 3.75E-04 |
| Forest damage | | | | 2.94E-05 | 2.94E-05 |
| Totals | | | | 1.35E-01 | 3.37E-02 |

calculated with EcoSense 1.03

**Former 3.29** Damages caused by the assembly of windows (upper table) and damages of the total process chain (lower table) under the old evaluation modus

| CRAFTSMEN | Acute mortality (mECU/kWh) | Chronic mortality (mECU/kWh) | Morbidity (mECU/kWh) | Sub-totals incl. chronic mortality Scenario I (mECU/kWh) | Sub-totals excl. chronic mortality Scenario I (mECU/kWh) |
|---|---|---|---|---|---|
| Health damage | | | | | |
| TSP | 1.07E-03 | 4.32E-03 | 2.47E-04 | 5.73E-03 | 1.41E-03 |
| Sulphate | 3.62E-04 | 1.34E-03 | 7.74E-05 | 1.78E-03 | 4.39E-04 |
| Nitrate | 1.12E-02 | 4.15E-02 | 2.39E-03 | 5.51E-02 | 1.34E-02 |
| Crop damage | | | | 0.00E+00 | 0.00E+00 |
| Material damage | | | | 1.33E-04 | 1.33E-04 |
| Forest damage | | | | 7.06E-06 | 7.06E-06 |
| TOTAL | | | | 6.27E-02 | 1.56E-02 |

**Former 3.29** continued

| TOTAL PRODUCTION | Acute mortality (mECU/kWh) | Chronic mortality (mECU/kWh) | Morbidity (mECU/kWh) | Sub-totals incl. chronic mortality Scenario I (mECU/kWh) | Sub-totals excl. chronic mortality Scenario I (mECU/kWh) |
|---|---|---|---|---|---|
| Health damage | | | | | |
| TSP | 8.44E-02 | 2.88E-01 | 1.66E-02 | 3.82E-01 | 9.41E-02 |
| Sulphate | 1.15E-01 | 4.27E-01 | 2.46E-02 | 5.67E-01 | 1.72E-01 |
| Nitrate | 2.43E-01 | 8.97E-01 | 5.17E-02 | 1.19E-00 | 2.95E-01 |
| Crop damage | | | | 0.00E+00 | 0.00E+00 |
| Material damage | | | | 7.77E-03 | 7.77E-03 |
| Forest damage | | | | 3.76E-04 | 3.76E-04 |
| Total | | | | 2.15E-00 | 5.37E-01 |

calculated with EcoSense 1.03

**Former Table 3.36** External costs due to the final assembly process of heat exchangers in the DIW scenario (upper table) and due to all production and pre-production processes of heat exchanger, production capacity expansion, and maintenance/modernisation (lower table) under the old evaluation modus

FINAL ASSEMBLY

| Health damage | Acute mortality (mECU/kWh) | Chronic mortality (mECU/kWh) | Morbidity (mECU/kWh) | Sub-total incl. chronic mortality (mECU/kWh) | Sub-total excl. chronic mortality (mECU/kWh) |
|---|---|---|---|---|---|
| TSP | 3.52E-04 | 1.31E-03 | 7.54E-05 | 1.73E-03 | 4.28E-04 |
| Sulphate | 3.15E-04 | 1.17E-03 | 6.74E-05 | 1.55E-03 | 3.82E-04 |
| Nitrate | 2.90E-03 | 1.08E-02 | 6.20E-04 | 1.43E-02 | 3.52E-03 |
| Crop damage | | | | 0.00E+00 | 0.00E+00 |
| Material damage | | | | 0.00E+00 | 0.00E+00 |
| Forest damage | | | | 2.59E-06 | 2.59E-06 |
| Total | | | | 1.76E-02 | 4.33E-03 |

TOTAL PRODUCTION

| Health damage | Acute mortality (mECU/kWh) | Chronic mortality (mECU/kWh) | Morbidity (mECU/kWh) | Sub-total incl. chronic mortality (mECU/kWh) | Sub-total excl. chronic mortality (mECU/kWh) |
|---|---|---|---|---|---|
| TSP | 1.62E-02 | 6.00E-02 | 3.46E-03 | 7.96E-02 | 1.96E-02 |
| Sulphate | 6.54E-03 | 2.43E-02 | 1.40E-03 | 3.22E-02 | 7.94E-03 |
| Nitrate | 1.96E-02 | 7.29E-02 | 4.20E-03 | 9.67E-02 | 2.38E-02 |
| Crop damage | | | | 0.00E+00 | 0.00E+00 |
| Material damage | | | | 5.02E-04 | 5.02E-04 |
| Forest damage | | | | 5.51E-05 | 5.51E-05 |
| Total | | | | 2.09E-01 | 5.20E-02 |

calculated with EcoSense 1.03

# 4. ANALYSIS OF COGENERATION PLANTS

## 4.1 Introduction

Combined Heat and Power (CHP) production offers a significant rise in fuel efficiency, leading to a decrease of environmental burdens emitted per unit of end-use energy. Combined production of heat and electricity is possible with all heat machines and fuels from a few kW-rated to large steam condensing power plants. Heat-plus-power efficiencies are typically in the range of 80 - 90%. The employment of combined heat and power production is closely linked with the availability of district heating networks and central heating systems in buildings. District heating networks are energy transmission systems suited for the distribution of heat within areas with sufficiently high heat load densities.

The objective of Task 2.2 "Analysis of Cogeneration Plants" was to integrate CHP-technologies into the existing ExternE accounting framework. While the same methodological framework that has been established for electricity generation systems can be applied for impact assessment, the allocation of damages between different products, namely electricity and heat, is a new methodological issue addressed within this task. Plants with different fuels (biomass, natural gas, coal) at different sites (Germany, Finland, Italy) operated under different conditions were selected in order to demonstrate the feasibility of the approach and to provide externality figures for a wide range of different CHP applications.

The following sections first develop an approach for the allocation of damage between electricity and heat, and then describe the implementation of the methodology in different case studies in Germany, Italy and Finland. The quantification of impacts and external costs follows the approach described in detail in chapter 2.

## 4.2 Allocation of external costs between electricity and heat

### 4.2.1 Options for allocating impacts between electricity and heat

The attribution of costs to either heat or electricity is a well know problem in the field of internal cost accounting, and thus a number of alternative approaches are discussed in the literature, providing a helpful starting point for the discussion on how to treat external costs. Some of the more important options are briefly described in the following:

- **Allocation based on operational characteristics**
  In many cases, a CHP-plant is operated to primarily satisfy a predefined heat or electricity demand, so that the additional production of respectively electricity or heat might be considered as a by-product. In this case it seems plausible to assign all the benefits of the

joint production to the by-product, i.e. giving a credit for the production of heat, if the plant is operated towards electricity production, and giving a credit for electricity generation, if the plant is operated towards the production of heat. The respective credits are determined by the damage costs avoided due to substituting other technologies. Taking into account the operational characteristics, we can also use the ratio of additional fuel input required for producing the 'by-product' to the total fuel input required for both electricity and heat production.

- **Allocation based on thermodynamic parameters**
  A simple and frequently used indicator is the energy content of the product streams, which is used as a weighing factor for the allocation of costs or impacts between electricity and heat. Taking into account the higher 'value' of mechanical or electrical energy compared to heat, the product's exergy is a reasonable additional indicator. Several other parameters are discussed in the literature, taking into account various characteristics of the steam-process, but results achieved with these parameters generally are within the range covered by the energy content and the exergy indicators, so that they are not further discussed here.

- **Allocation based on the final products' price**
  Assuming a perfect market, the final products' price probably is the best indicator for the 'utility' of the product. However, the price of electricity and heat is highly dependent on the customers demand characteristics and other non-technical parameters.

The respective advantages and disadvantages are briefly summarised below:

- **Credit for heat/electricity**

  + Commonly used and accepted method for the calculation of internal costs.

  - Results are highly dependent on the operational context of the power plant itself, and of the substituted technology which is used as reference for the calculation of credits.

  ? There are at least some doubts about consistency with the general ExternE-approach when avoided costs due to substituting other technologies are included in the calculation of damage costs.

- **Thermodynamic parameters**

  + The allocation based on thermodynamic parameters is clearly defined by the power plant's characteristics and does not depend on the context.

  ? The allocation based on thermodynamic parameters as such is not very meaningful. Results are difficult to interpret with regard to energy/economy problems.

- **Price**

  + The price of the final products is probably the best indicator for the 'utility' of the products and thus an appropriate indicator for allocation.

  - The price is dependent on the power plant's operational context as well as on non-technical market factors.

The thermodynamic parameters are the only ones that allow a consistent comparison between sites and between different CHP-technologies without being influenced by the respective non-technical conditions. As far as prices as the probably best measure of 'utility' are well defined for both product streams, the allocation based on the price indicator certainly provides additional useful information.

### 4.2.2 Allocation of external costs based on thermodynamic parameters

Among the alternative approaches to compare the relative amounts electricity and heat produced by a cogeneration plant the thermodynamic measures are best defined, but even they are far from being unique. The first and most important choice is the selection of the basic thermodynamic quantity to use in the comparison. The two choices to be considered further are energy (or more precisely enthalpy) and exergy.

Independently of whether enthalpy or exergy is used as the measure of production, the energy streams to be considered and the points of measurement have to be specified. The following choices are recommended for application unless exceptional circumstances make some other choice clearly preferable:

- the measurements are to be made at the plant periphery taking into account both outgoing and incoming streams, more specifically,
- net production is used for electricity,
- heat production as well as the temperature and pressure are measured at the exit from the site,
- the energy or exergy content of the return flow has to be calculated similarly and subtracted from the exit flow.

***Allocation based on energy flows***

In the case of energy the amount of electricity is in most cases unambiguously known. The enthalpy of the heat energy sent out is also unambiguous in principle, but here may be some practical difficulties, which must be solved in practice. Typical problems include:

- The properties of the return flow are not always immediately available. This may be a problem for steam producing plants, where the production is commonly specified in terms of steam properties and mass flow. When the production is given in energy units, the return flow is in general already subtracted.

- Many plants supply steam at several different pressure levels or both steam and hot water. Occasionally the return flow may also occur at two different temperature levels. Additional care is needed to calculate the net enthalpy supply in such situations.

In most cases the annual production figures are directly available and represent the net enthalpy production. In this case the details of the process are irrelevant to the present case.

***Allocation based on exergy flows***

The most uniformly applicable approach to determining the exergy contents of energy flows produced by an energy production unit is to determine first the enthalpy flows and the ratios of exergy to energy for each flow. The exergy flows are then obtained as product of these factors.

For electrical energy exergy equals enthalpy. Thus no further calculations are needed. In the case of heat the situation is significantly more complicated.

The most important single choice in determining the exergies is the selection of the environmental temperature required by the definition of exergy. As the calculation concerns the exergy content of the heat production, the natural reference point is the typical outside temperature. For district heating applications it is natural to emphasise the heating period. In industrial applications the yearly average temperature could be used. For practical reasons it is proposed that the following values are to be used: +20 $^{o}$C for Southern Europe, 0 $^{o}$C for Sweden, Finland and Norway and +10 $^{o}$C for other countries.

For liquid water the density variations have an insignificant effect on exergy. Furthermore the specific heat of liquid water varies by less than 1 % between 0 °C and 100 °C and increases only slowly at higher temperatures (under pressure). Neglecting the variations of density and specific heat the ratio of exergy X to enthalpy H can be calculated from the formula:

$$X/H = 1 - (T_0/(T_{out} - T_{in})) \ln (T_{out}/T_{in})$$

where temperatures are $T_{out}$ for the outgoing flow, $T_{in}$ for the incoming return flow and $T_0$ for the environment. The temperatures must be given as absolute temperatures in Kelvin.

In the more general case, where steam is being produced, the exergies of all separate outgoing and incoming streams must be determined and the net exergy flow determined from these. The general formula for calculating the specific exergy of a stream is

$$x = h - h_0 - v(p - p_0) - T_0(s - s_0),$$

where the quantities $h_0$, $p_0$, $T_0$, and $s_0$ refer to the corresponding parameters under selected environmental conditions. h is specific enthalpy, v specific volume (equal to inverse density), p pressure and s specific entropy. The only practical way to obtain the values of the thermodynamic parameters is the use of steam tables or computer programs producing equivalent data. A small easy-to-use Windows application for obtaining this information was provided for the ExternE work by the VTT Energy group.

## 4.3 The German case studies

### 4.3.1 Description of reference technologies

For the German case study, two different technology options are analysed:

- a small size combined cycle gas fired CHP plant, and
- a large coal-fired CHP plant, equipped with dust precipitators, FGD and DENOX facilities.

While the technical characteristics of the reference plants correspond to hypothetical power plants described in the IKARUS database, which was developed in Germany to provide a comprehensive database for analysing instruments for greenhouse gas emission mitigation strategies, we have selected sites where existing plants of comparable size are operated, i.e. where a district heat network is available. For the gas fired plant we assume a site in the city of Stuttgart, while the large coal fired power plant is located near the city of Mannheim.

For the quantification of burdens from the upstream processes (fuel extraction and preparation, fuel transport), data on the coal and gas fuel cycles for electricity generation are used that are described in the German National Implementation report.

***Combined cycle gas fired CHP plant***

The reference plant to be analysed is a natural gas fired combined cycle plant with condensing turbine. The maximum heat production is 145 MJ/s at simultaneous electricity production of 165 $MW_{el}$. Operation at rated load (live steam conditions of 508 °C/66bar) result in a thermal efficiency and utilisation ratio of 47.0% and 88.3% respectively, which might decrease at different operational modes (e.g. part load operation, different ratios of electricity/heat production). The heat flow temperature is 90 °C. Some of the technical characteristics are summarised in Table 4.1, Figure 4.1 shows the system flowsheet of the process.

***Coal fired CHP plant***

The reference plant is an atmospheric coal dust fired plant with condensing turbine, equipped with electrostatic dust precipitators, FGD and DENOX facilities. The maximum heat production is 750 MJ/s at simultaneous electricity production of 523 $MW_{el}$. Operation at rated load (live steam and superheating conditions of 540 °C/250 bar and 560 °C/56.0-51.1 bar) result in a thermal efficiency and utilisation ratio of 37.5% and 91.3% respectively. Again, different operational modes lead to a decrease of thermal efficiency and utilisation ratio. The technical characteristics most relevant for impact assessment are summarised in Table 4.1, and Figure 4.2 shows the system flowsheet of the process.

***Emissions from upstream processes***

Emissions from upstream processes are quantified following the assumptions described in detail in the coal fuel cycle report (European Commission, 1995) and in the German National Implementation report (Krewitt et al., 1995). Coal for the Mannheim CHP plant is produced in

**Table 4.1** Technical characteristics of the reference power plants

| | Gas CC | Coal |
|---|---|---|
| *Capacity* | | |
| electricity | 165.1 $MW_{el}$ | 575.5 $MW_{el}$ |
| heat | 145.0 MJ/s | 750.0 MJ/s |
| thermal efficiency | 47.0 % | 37.5 % |
| utilisation ratio | 88.3 % | 91.3 % |
| *Annual production* | | |
| electricity | 429.6 E+6 $kWh_{el}$/a | 2026.2 E+6 $kWh_{el}$/a |
| heat | 1358.2 E+9 kJ/a | 10454.4 E+9 kJ/a |
| Full load hours per year | 2602 h/a | 3872 h/a |
| Fuel input | 3287.9 E+12 J/a | 19436.8 E+12 J/a |
| *Fuel specification* | | |
| Calorific value of fuel | 50.06 MJ/kg | 29.2 MJ/kg |
| Sulphur content | | 0.9 % |
| *Emissions* | | |
| $SO_2$ | - | 666 t/a |
| $NO_x$ | 140 t/a | 666 t/a |
| TSP | - | 100 t/a |
| $CO_2$ | 174000 t/a | 1815000 t/a |
| *Data relevant for air quality modelling* | | |
| Stack height | 170 m | 240 m |
| Stack diameter | 6 m | 10 m |
| Flue gas volume stream | 1.079 E+6 $Nm^3/h$ | 1.72 E+6 $Nm^3/h$ |
| Flue gas temperature | 115 °C | 130 °C |

**Table 4.2** Emissions from upstream processes

| | Coal CHP plant | Gas CHP plant |
|---|---|---|
| $SO_2$ (t/a) | 88.2 | 1.6 |
| $NO_x$ (t/a) | 78.9 | 36.3 |
| TSP (t/a) | 8.1 | 9.5 |
| $CO_2$ (kt/a) | 78.9 | 7.9 |

underground mines at the German Ruhrgebiet and Saarland coal fields. We assume that about 60% of the coal is transported by barge, and the remaining fraction by rail to the power plant site. The quantification of emissions from the upstream processes of gas supply for the Stuttgart plant is based on the typical German gas mix, using about 26% domestic gas and imports from the GUS, the Netherlands, Norway and Denmark. The resulting emissions are summarised in Table 4.2.

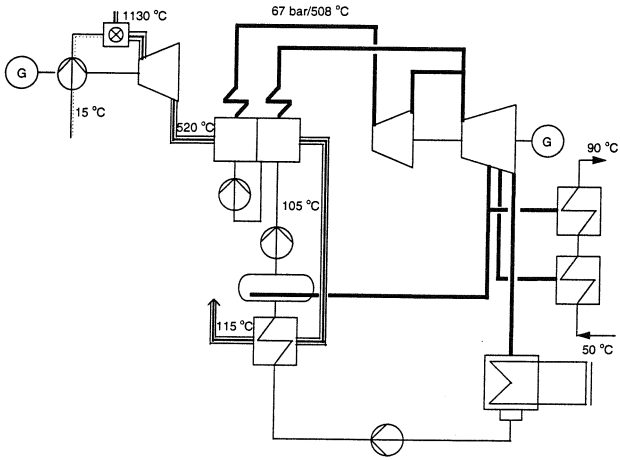



**Figure 4.1** Flowsheet of the gas fired reference power plant

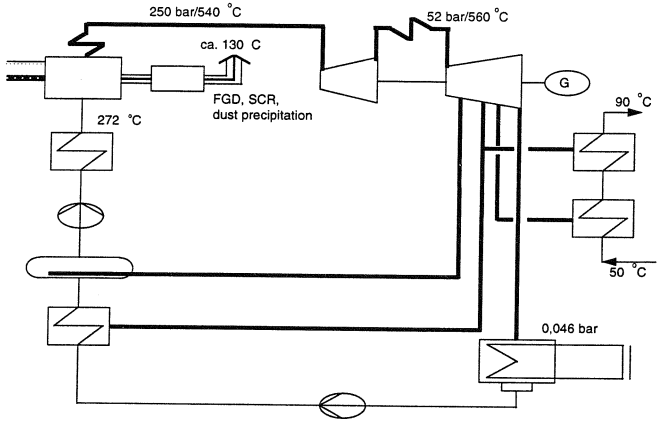



**Figure 4.2** Flowsheet of the coal fired reference plant

### 4.3.2 Quantification of damage

***Damage from airborne pollutants***

The quantification of impacts and damages follows the approach described in detail in chapter 2. The priority impact pathways analysed are the same as those identified for the coal and the gas fuel cycle, including effects on human health, crops, building materials and climate change. Except for global warming, the quantification of impacts is based on EcoSense runs (Version 2.0). For the valuation of mortality impacts we have applied both the *Value of Statistical Life (VSL)* approach and the *Value of Life Year Lost (VLYL)* approach, using the 3% discount rate. As sensitivity analysis, Table 4.4 shows the resulting annual damage costs for different discount rates. To quantify damage resulting from greenhouse gas emissions, we use the 95% confidence range of 3.8–139 ECU per tonne of $CO_2$ and the illustrative restricted range of 18–46 ECU/t $CO_2$ recommended by the *Global Warming Task Group* of the ExternE Project (see Appendix VII).

Emissions from upstream processes are added to the power plant's emissions in order to derive a first estimate of damages from the full *life cycle*. The resulting annual damage costs are shown in Table 4.3. Similar to other fossil fuel cycles, health impacts clearly dominate the overall damage costs. Besides of the impact categories quantified, there exist a number of impacts like e.g. damage to ecosystems, impacts from transmission lines etc. that we were not yet able to quantify adequately in ExternE.

***Occupational health impacts***

Occupational health impacts are mainly assessed by evaluating occupational health statistics, following the approach described in detail in the ExternE Methodology report as well as in the German National Implementation report. To quantify impacts from natural gas extraction, data from the UK and from Norway are used. Although data from the two countries differ considerably - partly because of different impact categories used - accidents rates averaged over both countries are used. Lung cancer from exposure to radon and coalworkers pneumoconiosis from exposure to dust are occupational impacts of major concern related to underground coal mining. As these diseases show a long term latency period, exposure-response functions (which are described in the Methodology report) are used to estimate effects resulting from current exposure levels.

Following the approach suggested in the German National Implementation report, we have calculated both the total and the *net* impacts (Table 4.5 and Table 4.6), but we have used the net risk to calculate the resulting damage costs. Due to the negative net risk of minor injuries from the gas fuel cycle - resulting from the low risk of power plant operation - the damage costs are negative, but very small (-2939.7 ECU/a). The damage costs resulting from the net occupational health impacts from the coal fuel cycle amount to 1.9 Million ECU per year.

**Table 4.3** Annual damage costs in Million ECU (3% discount rate)

| | Coal | | Gas CC | |
|---|---|---|---|---|
| | power plant | upstream processes | power plant | upstream processes |
| Health | | | | |
| mortality | | | | |
| *VSL approach* | 48.3 | 4.5 | 4.6 | 1.7 |
| *VLYL approach* | 19.0 | 2.3 | 2.1 | 0.77 |
| morbidity | 2.6 | 0.32 | 0.33 | 0.12 |
| Crop losses | 0.13 | 0.016 | 0.028 | 0.0071 |
| Material impacts | 0.28 | 0.035 | 0.022 | 0.0065 |
| Global warming | | | | |
| 95% confidence interval | 6.9-252.3 | 0.3-11.0 | 0.7-24.2 | 0.03-1.1 |
| illustrative restricted range | 32.7-83.5 | 1.4-3.6 | 3.1-8.0 | 0.1-0.4 |
| Sub-total | | | | |
| *VSL approach* | | | | |
| 95% confidence interval | 55.6-301.0 | 4.9-15.5 | 5.3-28.8 | 1.7-2.8 |
| illustrative restricted range | 81.4-132.2 | 6.0-8.2 | 7.8-12.7 | 1.9-2.1 |
| *VLYL approach* | | | | |
| 95% confidence interval | 26.3-271.7 | 2.7-13.3 | 2.8-26.3 | 0.8-1.9 |
| illustrative restricted range | 52.1-102.9 | 3.8-6.0 | 5.3-10.2 | 0.9-1.1 |

**Table 4.4** Annual damage costs from health effects for different discount rates in Mill. ECU (VLYL approach, power plant and upstream processes)

| | Coal | | | Gas | | |
|---|---|---|---|---|---|---|
| | 0% | 3% | 10% | 0% | 3% | 10% |
| Mortality | 34.6 | 21.3 | 15.2 | 4.7 | 2.9 | 2.1 |
| Morbidity | 2.9 | 2.9 | 2.9 | 0.44 | 0.44 | 0.44 |
| Total | 37.5 | 24.2 | 18.2 | 5.1 | 3.3 | 2.5 |

### 4.3.3 Allocation of external costs to electricity and heat production

The annual damage costs presented in the previous sections are transformed to external costs per unit energy output below using both the exergy and the price indicator for allocation.

**Table 4.5** Occupational health impacts from the gas reference fuel cycle (cases per year)

| | fatalities | | major injuries | | minor injuries | |
|---|---|---|---|---|---|---|
| | total | net | total | net | total | net |
| gas extraction | 0.0024 | 0.0017 | 0.036 | 0.020 | 0.33 | - 0.35 |
| power plant operation | 0.0017 | - 0.0021 | 0.038 | - 0.050 | 1.8 | - 1.9 |
| construction a. dismantling | 0.0037 | 0.00050 | 0.11 | 0.035 | 4.2 | 1.2 |
| Total | 0.0077 | 0.000037 | 0.18 | 0.0052 | 6.3 | - 1.1 |

**Table 4.6** Occupational health impacts from the coal reference fuel cycle (cases per year)

| | fatalities | | major injuries | | minor injuries | |
|---|---|---|---|---|---|---|
| | total | net | total | net | total | net |
| mining | 0.43 | 0.36 | 19.9 | 18.4 | 143 | 77 |
| limestone extraction | 0.007 | 0.0050 | 0.14 | 0.10 | 3.8 | 1.9 |
| coal transport | 0.054 | 0.046 | 0.34 | 0.15 | 10.9 | 3.1 |
| other transport | 0.017 | 0.011 | 0.21 | 0.072 | 6.5 | 0.73 |
| power plant operation | 0.0053 | - 0.0067 | 0.12 | - 0.16 | 5.6 | - 6.1 |
| construction a. dismantling | 0.016 | 0.0022 | 0.47 | 0.15 | 18.5 | 5.2 |
| Total | 0.53 | 0.42 | 21.2 | 18.7 | 188.6 | 81.9 |

***Allocation based on exergy***

While for electrical energy exergy equals enthalpy, the ratio of exergy (X) to enthalpy (H) for heat is calculated according to the following equation:

$$X/H = 1 - (T_o/(T_{out} - T_{in})) \ln (T_{out}/T_{in})$$

with $T_{out}$ temperature of the outgoing flow
$T_{in}$ temperature of the incoming return flow
$T_o$ temperature of the environment

For the two reference plants we use $T_{out}$ = 363 K (90 °C), $T_{in}$ = 323 K (50 °C) and $T_o$ = 283 K (10 °C), leading to the exergy to enthalpy ratio of 0.174 . The total annual exergy production is

*for the Gas CC CHP plant:*

$$1.5 \text{ E+15 J} \times 1 + 1.36 \text{ E+15 J} \times 0.174 = 1.74 \text{ E+15 J}$$

*for the Coal CHP plant:*

$$7.29 \text{ E+15 J} \times 1 + 1.045 \text{ E+16 J} \times 0.174 = 9.1 \text{ E+15 J}$$

**Table 4.7** Damage costs from the gas fired reference CHP plant

| | Damage per year in Mill. ECU | Allocation based on Exergy $T_{out}$ = 363 K $T_{in}$ = 323 K $T_o$ = 283 K | | Allocation based on Price Electricity: 19 Pf/kWh Heat: 20 DM/GJ | |
|---|---|---|---|---|---|
| | | mECU/kWh$_{el}$ | mECU/MJ | mECU/kWh$_{el}$ | mECU/MJ |
| Public health mortality | | | | | |
| *VLYL approach* | 2.9 | 5.7 | 0.29 | 5.0 | 0.53 |
| *VSL approach* | 6.3 | 12.6 | 0.65 | 11.0 | 1.2 |
| morbidity | 0.44 | 0.88 | 0.045 | 0.77 | 0.081 |
| Occupational health | -0.0029 | -0.0059 | -0.00031 | -0.0051 | -0.00054 |
| Crops | 0.035 | 0.070 | 0.0036 | 0.061 | 0.0064 |
| Materials | 0.029 | 0.058 | 0.0030 | 0.050 | 0.0053 |
| Global warming | | | | | |
| 95% confidence interval | 0.7-25.3 | 1.4-50.7 | 0.07-2.5 | 1.2-44.2 | 0.1-4.6 |
| illustrative restricted range | 3.3-8.4 | 6.6-16.8 | 0.3-0.8 | 5.7-14.6 | 0.6-1.5 |
| Sub-total | | | | | |
| *VSL approach* | | | | | |
| 95% confidence interval | 7.1-31.6 | 14.2-63.5 | 0.7-3.2 | 12.3-55.3 | 1.3-5.8 |
| illustrative restricted range | 9.6-14.7 | 19.3-29.6 | 1.0-1.5 | 16.8-25.7 | 1.8-2.7 |
| *VLYL approach* | | | | | |
| 95% confidence interval | 3.6-28.2 | 7.3-56.6 | 0.4-2.8 | 6.3-49.3 | 0.7-5.2 |
| illustrative restricted range | 6.2-11.3 | 12.5-22.7 | 0.6-1.1 | 10.8-19.7 | 1.1-2.1 |

Thus, in the case of the gas fired plant, 86% of the damage is allocated to the production of electricity and 14% to the heat production, while in the case of the coal fired plant 80% of the damage is allocated to electricity and 20% to heat production.

***Allocation based on price***

For the allocation based on price we use the average price for electricity in Germany of 19 Pf/kWh, and an average price for district heat for domestic buildings of 20 DM/GJ. Based on these prices, the total value of the annual electricity and heat production amounts to

*for the Gas CC CHP plant:*

429.6 E+6 kWh x 0.19 DM/kWh + 1358.2 E+3 GJ x 20 DM/GJ = 108.8 Mill. DM

## 4.4 The Italian case study

### 4.4.1 Plant description

This case study deals with the external costs of an industrial Combined Heat and Power (CHP) plant and the allocation of its impacts between electricity and heat produced. A case of industrial cogeneration has been chosen in Italy with high electricity to heat ratio, as for the independent power producers aimed at supplying electricity to the grid in the liberalised industry. Its efficiency is quite good and the heat demand rather flat during the day and the year. The plant is located in a highly industrialised area where heat demand and fuel supply are available.

***The fuel cycle***

The whole fuel cycle of the reference plant is quite long: the gas burnt in Italy comes mainly from the Algerian fields via the trans-mediterranean pipeline that connects Tunisia and Sicily. The other sources are Russia and The Netherlands, in a lesser extent. Our plant is directly connected to the high pressure gas grid. Very few information is available on the drilling and operating conditions in the Sahara desert as well as on the gas pipe operation along the many hundred kilometres in African countries. We are thus inclined to consider the impact of the upstream phase of the cycle simply adopting the results achieved in the previous ExternE oil and gas study (EC 1995, vol. 4). Downstream the power generation stage, instead, the fuel cycle is quite short, as very few wastes are released from the plant.

***The reference plant***

The reference CHP plant has been chosen as a significant example of those under construction in the nineties in Italy and in Europe where independent generation is allowed. It has been built in an existing chemical plant that produces basic chemical products in a simple, but energy intensive, process. The steam consumption of the chemical plant varies according to the mix of products, ranging from 220 to 465 t/h globally. Nevertheless, for more than 50% of the time the steam demand is some 405 t/h and this is our reference figure. All the produced electricity, net of the CHP self consumption, is sold to the grid and the chemical plant purchases its own consumption from the grid.

The ratio between electricity and heat generation is quite high, in order to maximise the income from the sale of electricity. The total gross electricity production is 365.5 MW, while the electricity sent out is 354.4 MW. The total heat supplied to the chemical plant is 286 MW, as shown in detail in Table 5.9. The overall efficiency of the plant, or fuel use factor, is 72.6%, while the electric efficiency alone is 40.7 %. Figure 4.3 shows the simplified scheme of the plant.

There are 2 gas turbines GT V94.2 of 150 MW each built by Ansaldo under Siemens licence. They are fed with air blown by the compressor C and with 93517 $Nm^3$ per hour of natural gas (897 MW) and produce 148.7 $MW_e$ each at nominal conditions. The flue gas stream, 1008 kg/s at a temperature of 551°C passes through two recovery boilers at 3 pressure levels (72,

**Table 5.9** Technical characteristics of the reference gas fired CHP plant

| | |
|---|---|
| Capacity | |
| gross electricity | 365.5 $MW_e$ |
| net electricity | 354.4 $MW_e$ |
| heat | 286 MJ/s |
| Full load hours per year | 8000 |
| Annual fuel input | $25.836 \cdot 10^{15}$ J |
| Annual production | |
| electricity | $2.835 \cdot 10^{9}$ kWh |
| heat | $8.2368 \cdot 10^{15}$ J |

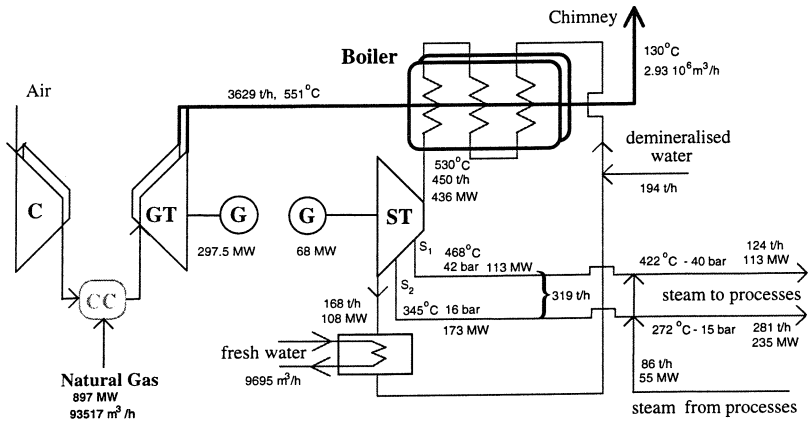



**Figure 4.3** Simplified scheme of the reference CHP plant

16, 2.4 bar), where steam is produced at 530°C (450 t/h) in a complete feed water regenerative heating cycle. High pressure steam feeds the steam turbine and supplies the chemical plant requirements. The steam turbine produces 68.12 $MW_e$ and 286 $MW_t$ of steam at two different pressure levels as required by the processes of the chemical plant. Part of the steam generated goes into the condenser where a 0.05 bar vacuum is created exchanging heat with river water. The steam circuit needs 194 t/h of demineralised water to integrate the flow returning from the processes. The total electric self-consumption of the auxiliary services of the CHP plant is 11 MW.

86 t/h of steam return from the processes and are mixed to the steam flow coming from the turbine. Table 4.10 shows the characteristics of thermal load at two different pressure levels. The utilisation factor of the plant is very high, as for energy intensive processes: the full load

**Table 4.10** Steam spilled from the turbine and exported to the processes

| | Spills | | Loads | |
|---|---|---|---|---|
| | 1 withdrawal | 2 withdrawal | High pressure | Low pressure |
| steam [t/h] | 120.4 | 198.5 | 124 | 281 |
| temperature [K] | 741 | 618 | 695 | 545 |
| pressure [bar] | 42 | 16 | 40 | 15 |
| enthalpy [kcal/kg] | 810 | 750 | 770 | 708 |
| [MW] | 113 | 173 | 110 | 231 |

hours are 8000, although some unexpected stop could diminish this figure. The electricity generated is 2835 GWh per year, while heat amounts to 8236.8 TJ.

The chemical plant was previously connected to the 150 kV grid and a new connection to a 380 kV line has been required by the grid operator Enel. In the list of the impacts should be also included the 4 km new high voltage line built, even if it is situated in an highly industrialised area and the amenity impact is negligible.

***Plant locations***

In order to evaluate the site sensitivity of the results we located the plant in two different sites: the main reference location is in the industrial area around Milan, the second one has been chosen in Brindisi in southern Italy, where the meteorological conditions are quite different. The comparison between two sites in the same country, but with different natural conditions can give some hints on the site specificity of the impacts and the opportunity to control the location of new plants. These two sites well represent extreme conditions of the European region, being typical of continental and Mediterranean Europe.

### 4.4.2 Quantification of burdens for each stage

Even if the CHP fuel cycle is rather long, as already said, we focus on the step interesting for the task assigned to this study, which is the allocation of impacts between heat and electricity in the combined production. We make thus reference to the previous studies of the ExternE project as far as the common parts of the impact pathway are concerned, like the upstream of the gas fuel cycle.

***Exploration, extraction and transmission of gas***

Missing detailed information on the drilling activity in the Sahara desert by the Algerian State owned company, we attribute to our fuel cycle the same impact calculated in the previous ExternE gas fuel cycle. Even if they have a minor weight, the main burdens of these activities are listed below:

- accidents and inconvenience caused by the civil works for well drilling and pipe construction: this is a very difficult figure to quantify, due to the remote location of the

wells; moreover, the plant consumption is only some 5% of the total gas imported from Algeria via pipelines and the upstream impact should be reduced accordingly;
- emissions of the vehicles operating during wells' drilling and pipe construction;
- chemicals discharged in the environment in exploration and development; when a well is drilled an oil based mud is used made of barite (60%), base oil (32%) and other inert components. Such a mud is only partially biodegradable;
- fugitive emissions of natural gas in transmission are evaluated around 3% of throughput on the base of "gas unaccounted for"; a lower figure is given by the industry (1%). As a considerable part of these losses are in the distribution network, not used by our plant, a 0.5% figure is chosen, higher than that chosen in the gas fuel cycle as a consequence of the long distance covered by the pipe (~2500 km).

***Plant construction***

During plant construction some temporary inconveniences can disturb the local activities. Such inconveniences are due to traffic increase (and thus accidents), noise, interference with the business of other factories (the area is highly industrialised), production of wastes, etc. Civil works are expected to last 30 months, with an average presence of 200 workers. The increase of the traffic is estimated negligible, considered the great quantity of trucks that already are in the area, apart from some temporary cases when exceptional loads have been transported in the site (turbines, transformers, other bulky goods). In this step the occupational health impact has its highest incidence, being the work-force in civil engineering the most exposed to the risk of serious accidents.

As far as the emissions of $CO_2$, $SO_2$, $NO_x$, and particulate caused by the construction of a fossil fuel plant are concerned, they are three orders of magnitude smaller than those from the operation, and obviously they are not worth the same effort of analysis as direct emissions.

***Plant operation***

During plant operation the major part of the fuel cycle impacts is produced. Stack emissions account for the single biggest figure in the total evaluation, in spite of the low pollutant content of the natural gas. Nitrogen oxide emissions are quite low thanks to the steam injection into the combustion chamber to reduce the temperature of the flame: the value of 47.75 mg/Nm$^3$ is well below the present limit set by the EU Directive on thermal power stations' emissions. Let alone the compounds released in very low quantities, $NO_x$ CO and $CO_2$ are the pollutants released from the stack that we take into account in our work.

It is interesting to note how in the chemical plant, given the heat demand, the global emission are reduced with the new CHP installation in spite of the new electric power generation. According to the data supplied by the constructor, there is a dramatic reduction in the global emissions, as shown in **Table 4.11**. The stack height is 40 m and the flue gas stream ($2.93 \cdot 10^6$ Nm$^3$/h) has a temperature of 405 K. Carbon dioxide emissions from plant operation are 4.93% in weight of the flue gas stream, i.e. some 49.6 kg/s, with an equivalent annual figure of 1.43 million tonnes. The quantification of the contribution to global warming is quite uncertain, but

**Table 4.11** The emission of the chemical plant compared before and after the construction of the CHP plant

| | before | after | |
|---|---|---|---|
| | t/y | t/y | ppm |
| $NO_x$ | 1400 | 1100 | 25 |
| CO | 603 | 254 | 9 |
| $SO_2$ | 5120 | 0 | 0 |
| TSP | 31 | 0 | 0 |

even with a cautious estimation its impact is sensible. Also at this stage there is a burden on the work-force of the plant, but less important than that of the construction stage.

***Plant dismantling***

At the end of the operational life of the plant the site has to be reconverted as it was before the plant construction. In our case the location inside an old industrial site reduce the constraint of the environment reconstruction. The only burden to be taken into account is the possibility of accidents to the work-force.

***Review of other impacts***

Other impacts are identified that can not be monetized with a credible accuracy. We just recall them to the attention of the reader, aware of their limited scope.

**Noise.** Noise levels have been accurately controlled in order to comply with the enforced laws. As a matter of fact, due to the location inside the chemical plant where the background noise is quite high, in the reference case this impact is a minor one. Present Italian regulations impose a level of 80 dB at 1 metre from any part and 50 dB at the plant border (D.L. 277/1991). The project foresees a level of 60 dB in the control room and an increase of the background level lower than 6 dB at 100 m from the area where the gas turbines are located.

**Visual impact.** The location inside the chemical plant area reduces also the visual disamenity: the CHP plant is less than 50.000 $m^2$ large, i.e. some one tenth of the total area of the industrial plant. Moreover, the highest part of it, the chimney 40 m high, is quite smaller than the existing evaporative towers of the chemical plant and almost disappears among them. An observer from outside the plant could hardly perceive any change. In addition the dismantling of 4 out of 5 old heaters will free a considerable amount of space.

**Impacts on freshwater ecosystem.** The water consumption of the plant is rather considerable and is mainly withdrawn from a river that runs nearby. A water flow of 9695 $m^3/h$, or 2.7 $m^3/s$, is required at the condenser to cool the steam circuit and create the back-pressure (0.05 bar). The water is released into the river with a temperature increase of 9°C at the outlet in normal conditions and it is not expected any effect on the river ecosystem, thanks to the fast cooling of the water when re-mixed to the natural flow.

The steam cycle also requires de-mineralised water to replace the losses along the cycle in the quantity of 60% of the total circulating steam. It is supplied by the chemical plant that already had a demineralisation plant for its own consumption which has only been enlarged.

**Impacts caused by power lines.** We said 4 km of new high voltage power line has been built to connect the plant to the national transmission grid. The construction of high voltage transmission lines is for sure a major problem for siting new power plants: the strong opposition of the population often makes the realisation of an over-head line a matchless undertaking. As a matter of fact the impact of the power lines is a function of the plant location more than of the fuel cycle; we outline at a glance the impacts imputed to the lines, although it is really difficult to quantify and even to identify them without uncertainty. The impacts we can identify are:

- land use and forest cutting (not in our case);
- visual amenity;
- traffic and workers disturb during construction;
- noise;
- risk of accident;
- electromagnetic fields.

The visual impact is probably the most consistent, but is strongly subjective and could be estimated locally with hedonic prices techniques. The third and forth impacts indicated are really low and not significant in most cases, while the risk of accident can be due to collision or electric discharge. The probability of a crash against the pylons or the overhead lines is comparable with that of other aerial constructions for both humans and birds; the probability of a high current discharge to the ground dangerous for persons is close to nihil.

The impact from electromagnetic fields is the most discussed. But in spite of the their fame, there is no scientific evidence that electromagnetic fields originated from 50 Hz power lines can give rise to leukaemia or other cancer illness. We are thus inclined to neglect this impact.

The evaluation of impacts from power lines is a quite new discipline. As far as we know, there is only one study based on Contingent Valuation on the visual impact of power lines, made by a Canadian group, but with survey data taken in both France and Canada. The full reference is at the end of the chapter. The data from this study have recently been applied to a particular UK transmission line project and an external cost of 0.12 mECU/kWh has been estimated (Nick Eyre, personal communication). However, two points should be made about this:

- The CVM study results are not really transferable so the uncertainty is very high. Although the technology used for line construction is the same in all over Europe (with minor differences), the perception could be much different from case to case, being function of the landscape value and the "environmental consciousness" of the inhabitants as well.
- It applies to an increment of power rather than the power system as a whole.

An upper bound to the impact of the power lines could be found in the cost required for putting them underground in insulated cables. As a matter of fact it is a very high cost (and the

higher the voltage the higher the cost) that would not ever be paid by an electric utility, but can give a ceiling cost to the monetisation of the impact.

With reference to the visual amenity great efforts are devoted to design new line structures with a reduced impact: this is the case of the "compact lines" that have been developed by ENEL in Italy with the following characteristics:

- short distance between pylons, in order to have the height reduced and minor conductors' oscillation;
- use of new materials and shapes to have smaller insulating chains from the top of the pylon;
- substitution of the lattice structure of the pylon with a more agile one, with smaller dimensions at the ground.

In spite of an investment cost increase of some 10-40%, the pylon height can be reduced of 30-40% as well as the corridor proportions (Cirillo 1994).

### 4.4.3 Evaluation of impacts

***EcoSense results***

A major part of the evaluation of impacts has been performed using the EcoSense software developed by IER. It has been used for the dispersion analysis of airborne pollutants, the calculation of impacts and the quantification of damages. A detailed description of this software package is given in chapter 2. The absence of sulphur in the fuel used reduces considerably the overall impact.

***Public health effect***

The impact on the health of the European population has been calculated using the dose-response functions suggested by the EcoSense package. This is by far the main impact of our CHP plant and probably also that calculated with the best confidence, being the dose response functions for human health the most studied. There is not any impact calculated at the local level, as the acid aerosols have effect only at the regional level. The values in Table 4.12 are the medium given in the EcoSense report with a 3% discount rate for YOLL. The major contribute to the impact is given by the chronic Years of Lost Life calculated with the dose-response function proposed by Pope et al. in 1995: some 80% of the damage comes from the evaluation of this impact.

***Impact of air pollution on crops***

The impact on agriculture is quite small according to our results: we have no direct $SO_2$ emissions and there is small evidence of $NO_x$ effects on crops. Results from EcoSense are summarised in Table 4.13.

The total damage of 3332 ECU per year has really a minor importance when compared to the health damage. We should not forget that the crops receptors database is rather incomplete

**Table 4.12** Impact on human health

| Receptor | Impact | Pollutant | Dose-Response Function | Monetary unit value [ECU] | Damage [MECU/ year] |
|---|---|---|---|---|---|
| above 65 yrs | congestive heart failure | CO | Schwartz/Morris | 7870 | 0.154 |
| above 65 yrs | congestive heart failure | Nit | Schwartz/Morris | 7870 | 0.0073 |
| adults | 'chronic' YOLL | Nit | Pope | 84330 | 13.407 |
| adults | Restr. activity days | Nit | Ostro | 75 | 0.413 |
| adults | chronic bronchitis | Nit | Abbey | 105000 | 1.136 |
| asthma adults | Bronchodilator usage | Nit | Dusseldorp | 37 | 0.0466 |
| asthma adults | cough | Nit | Dusseldorp | 7 | 0.0091 |
| asthma adults | Lower resp. symptoms | Nit | Dusseldorp | 7.5 | 0.0035 |
| asthma children | Bronchodilator usage | Nit | Roemer | 37 | 0.0093 |
| asthma children | cough | Nit | Pope/Dockery | 7 | 0.003 |
| asthma children | Lower resp. symptoms | Nit | Roemer | 7.5 | 0.0025 |
| children | chronic cough | Nit | Dockery | 225 | 0.0433 |
| children | case of chr. bronchitis | Nit | Dockery | 225 | 0.0034 |
| human - total | resp. hosp. admission | Nit | Dab | 7870 | 0.0063 |
| human - total | resp. hosp. admission | nox | de Leon | 7870 | 0.015 |
| human - total | cerebrovascular hosp. adm | Nit | Wordley | 7870 | 0.0154 |
| TOTAL | | | | | 15.3 |

**Table 4.13** Estimate of the additional lime needed and of its cost as a consequence of the reference plant operation

| Receptor | pollutant | Dose-Response Function | Impact [t] | Monetary unit value [ECU/t] | Damage [ECU] |
|---|---|---|---|---|---|
| total crops | acid aerosols | CEC 1993 | 270.5 | 17.1 | 4626 |
| total crops | ncd | Hornung 1997 | -75.7 | 17.1 | - 1294 |

and most of the Mediterranean crops are not taken into account. Nevertheless, the minor incidence of this impact let us say that, even neglecting the effect on these crops, the final result is not substantially altered.

***Impact on building materials***

This is also a difficult figure to calculate. A further problem is the quantification of damage on artistic buildings that are so common in Europe and so exposed to airborne pollutants. This aspect is totally neglected in our case and even the calculation of impact over common houses is partial because of the problems related to the construction of the receptors database in a country, like Italy, where the building materials and typologies are so many. The output of EcoSense run gives a total damage of 0.136 MECU per year.

*Ozone impact*

The impact of Ozone pollution has been estimated according to the approach recommended by Rabl and Eyre in 1997, that gives 1500 ECU 95 per tonne of $NO_X$ emitted. 1100 ton released from the stack give an impact of 1.65 MECU per year.

*Occupational impacts*

**Impacts on health during construction.** We consider in detail the expected occupational impacts related to the construction of the power station, even if partially internalised. To evaluate occupational impacts arising from accidents during construction we followed a kind of macro-approach: we estimate the investment needed to build the power station, and then we multiply this figure by statistical data concerning average accidents per billion ECU1995 spent in the construction industry. Decommissioning risks are estimated in the same way of construction risks, on the basis of investment figures and can be reasonably be estimated in the region of 5 % of the total costs.

The construction of the plant implies an investment of about 340 MECU 1995, according to the present market prices. Dismantling costs have been estimated 17 MECU, therefore producing a total investment of 357 MECU 1995, with a 0% discount rate.

Italian statistics (INAIL 1980-82), regarding an average number of different accidents for billion ECU spent in construction, give the following coefficients:

| | |
|---|---|
| 5.95 | accidental death cases per billion ECU1995 |
| 128.5 | permanent damages to workers per billion ECU1995 |
| 1974.2 | temporary damages to workers per billion ECU1995. |

A permanent damage is defined as any kind of irreversible invalidity arising from any labour accident, whereas a temporary damage is considered an absence from work lasting more than three days, caused by any accident. The Value of Statistical Life is 3.1 MECU, "major injuries" have a value of 25000 ECU (UK Dept of Transport), while "minor injuries" 510 ECU (Markandya and Rhodes, 1992). This value has to be divided by the number of years of operation to obtain the impact per year (400 kECU). Therefore the resulting expected impacts on occupational health caused by labour accidents are represented in Table 4.14.

**Occupational impacts during operation.** To assess the impact on manpower health during plant operation, we follow the statistical approach already chosen for the construction. We make now reference to the statistical data used in the oil fuel cycle (EC 1995) shown in Table 4.15 and apply these figures to our reference case, where some 40 persons are employed, split

**Table 4.14** Impacts on workers' health during construction and dismantling

| Overall monetary impacts (MECU 95) | |
|---|---|
| 6.5 | accidental death cases |
| 1.15 | permanent damages to workers |
| 0.362 | temporary damages to workers |
| 8.012 | Total damages during construction and dismantling |

**Table 4.15** Occupational health risks during power plant operation

| | Management and services | Electrical engineering |
|---|---|---|
| Fatal accidents/person-year | 11.8E-6 | 25.9E-6 |
| Fatal diseases/person-year | 0.261E-6 | 8.46E-6 |
| Major accidents/person-year | 4E-4 | 7E-4 |
| Major diseases/person-year | 3.83E-6 | 70E-6 |
| Minor accidents/person-year | 0.021 | 0.026 |
| Minor diseases/person-year | 2E-4 | 13E-4 |

**Table 4.16** Damage costs from occupational health risks

| | ECU per year |
|---|---|
| Fatalities | 2878 |
| Major accidents | 587 |
| Minor accidents | 495 |
| TOTAL | 3960 |

in 4 turns during the 24 hours. We make the hypothesis that half of them work in the category management and services and half in the electrical engineering one.

With 20 person-year in each category and the given values for injuries, the damage cost of the occupational health risk are summarised in Table 4.16.

***Impact of global warming***

The main source of Greenhouse Gases (GhG) in the fuel cycle is by far the combustion in the power plant. The carbon dioxide emissions from the stack are 4.93% in weight of the flue gas stream, i.e. some 49.6 kg/s, with an equivalent annual figure of 1.43 million ton. According to the literature (EC 1995, N.4) the contribution of the upstream to GhG emissions is in the range of 2% of the power generation figure. The $CO_2$ emitted is thus some 1.46 Mton per year, while the other GhG emissions during plant operation ($N_2O$, $CH_4$) are neglected.

To assess the global warming damages the results of the sub-task group of EXTERNE on global warming are employed (see Appendix VII). Accordingly, the annual damage caused by our plant stands in the range of 5.5 to 202.9 million ECU(1995) (95% confidence interval). This is a rather broad estimate and let uncertain the impact of global warming. In terms of damage per unit electricity produced the range is 2–57 mECU/kWh.

***Estimation of the total impact***

We now proceed to make the sum of the different impacts caused by our fuel cycle, in order to have a final figure. All the impacts that have been mentioned only in qualitative terms are not reported here and a major uncertainty distinguishes this overall impact: the uncertainty of each single impact, in fact, is in some sense added to all the others.

**Table 4.17** Summary of the calculated impacts per year

| | Damage per year in MECU |
|---|---|
| Public health | 15.3 |
| Ozone | 1.65 |
| Crops | 0.0033 |
| Materials | 0.136 |
| Occupational health | 0.404 |
| Sub-total | 17.51 |
| Global warming (95% confidence interval) | 5.5–202.9 |
| **TOTAL** | 23.0–220.4 |

**Table 4.18** Comparison of the impacts in the two locations

| Damage per year in MECU | MILAN | BRINDISI |
|---|---|---|
| Public health | 15.3 | 9.59 |
| Mortality | 13.4 | 8.41 |
| Crops | 0.0033 | 0.0034 |
| Materials | 0.136 | 0.096 |
| **TOTAL** | 15.46 | 9.69 |

***The comparative site impacts***

In order to know the site-dependency of our results we have supposed to locate our plant in a different site to see how the impact changes in function of the new location. The technical data of the plant are exactly the same and the same considerations made about the qualitative impacts apply: the only difference is thus the result of the EcoSense run. The site chosen, as already said, is Brindisi in the south-eastern margin of the Italian peninsula. The impacts on occupational health, as well as on global warming and ozone remain unchanged.

The difference in the results is important: the impact of the airborne emission of the plant located in the peripheral location, surrounded by the sea is some 40 % lower. Nevertheless, the omission of the extra-European impact could explain part of this major variation.

### 4.4.4 The allocation of impacts and the evaluation of damages

A crucial task in this work is the allocation of impacts between the two products of the CHP plant: in this part we use different methodologies to split the impact calculated and we make a comparison of these approaches. This effort is important in order to allocate to the electricity production its own share of the global impact and to compare the externalities associated to this electricity generation technology with those of the traditional electric power stations.

***Allocation of impacts with different approaches***

The global impact of the plant, as well as each of its components, can be divided in many ways when the share attributable to the electricity production has to be found. We follow here three different approaches, according to

- energy content
- exergy content
- prices of products

**Energy.** The division of the total impact according to the energy content is the most immediate approach. Nevertheless the different *value* of the final products, either from the thermodynamic or the commercial point of view, is neglected and the heat produced can suffer an unfair allocation.

In our case the figures are the following:

Total energy production = 365.5 $MW_e$ + 286 $MW_t$ = 651.5 MW
Electricity share $_{energy}$ = 56.1 %
Thermal share $_{energy}$ = 43.9 %

The total damage without global warming according to this approach is 3.5 mECU/$kWh_e$ and 0.9 mECU/MJ.

**Exergy.** A different figure is obtained when the exergy, or *useful energy*, content of products is considered. The product streams' exergy is characterised by the respective Carnot-factors, calculated according to the steam tables. While the Carnot-factor for electricity is 1, the coefficients for the exergy to enthalpy ratios, function of the thermodynamic status of the two steam flows are the following (see Figure 4.3):

$\eta_{c1}$ = 0.339; $\eta_{c2}$ = 0.3027;

the thermal energy produced per year (8000 hours) is:

$E_{S1}$ = 113 MW = 3.254 E+15 J
$E_{S2}$ = 173 MW = 4.982 E+15 J

The total annual exergy production is:

(10.2067 E+15 J) x 1 + (3.2544 E+15 J) x 0.339 + (4.982 E+15) x 0.303 = 112.818 E+15 J

It yields that 79.6% of the damage is attributed to electricity generation, and 20.4% to heat production.

It can be useful to remember here that in the main Italian chemical group, that is also the main CHP autoproducer, uses to allocate operating costs between heat and power in the internal accounts according to the exergy criterion.

**Prices.** Another approach is to split the impact according to the prices of the final products. In our reference case, we consider the 1996 prices for electricity and steam for an Italian industrial user like our studied chemical plant.

The average price of *electricity* is calculated taking into account the two prices, peak hours and empty hours, paid in Italy to the independent producers. With 3500 peak hours out of 8000, the average price of the produced electricity is 97 ITL/kWh, or 50.8 mECU/kWh. The total annual value of the electricity is 144.02 MECU.

The *steam* price is more complicated to set and is evaluated as follows: as an alternative, steam would be produced in old boilers, already depreciated, i.e. with only variable costs. It is common practice in the industry to give a value to the steam taking into account just the fuel cost. This is the approach we also follow. A customer with 200 $Mm^3$ per year, or 25330 $m^3$ per hour over 8000 hours (as is our reference user when has to supply some 290 $MW_t$ to the processes), pays some 240 $ITL/m^3$ = 125 $mECU/m^3$, i.e. 7.8 ITL/MJ = 4.08 mECU/MJ. With an average efficiency of 90%, 8.63 $kWh_t$ of steam are produced in a steam boiler with one $m^3$ of natural gas. The value of each $kWh_t$ of steam is thus 14.66 mECU.

The total value of the output is given by the sum of electric and heat production. A kWh of electricity is valued 50.8 mECU, while a thermal kWh 14.66 mECU. According to the annual productions, the total value of the produced energy is 177.6 MECU. Following the criterion of prices, the share of electricity is 81.1%, the share of heat 18.9%. This is confirmed by the cash flow analysis of the CHP projects, where 3/4 of the turnover is expected to come from the sale of electricity.

***Quantification of monetary damages***

We have set up the criteria for the allocation of impacts and defined the means to calculate the damages attributable to both productions. In the following Table 4.19 the results obtained from the different methodologies are compared and the damages are calculated. It is interesting to note that almost 90% of the impact comes from only two figures, which are also those affected by the highest uncertainty.

**Table 4.19** Summary of results and comparison among allocation criteria

| | Damage per year in MECU | Allocation based on Exergy $T_{process\ heat}$ = 741 K & 618 K $T_{ambience}$ = 288 K | | Allocation based on Price Electricity: 50.8 $mECU/kWh_{el}$ Heat: 4.08 ECU/GJ | |
|---|---|---|---|---|---|
| | | $mECU/kWh_{el}$ | $mECU/MJ_{heat}$ | $mECU/kWh_{el}$ | $mECU/MJ_{heat}$ |
| Public health | 17.0 | 4.8 | 0.42 | 4.8 | 0.39 |
| Crops | 0.0033 | 0.0009 | 8.2 E-5 | 0.001 | 7.6E-5 |
| Materials | 0.14 | 0.038 | 0.0033 | 0.039 | 0.003 |
| Occupational health | 0.40 | 0.11 | 0.01 | 0.12 | 0.009 |
| Global warming | 5.5–202.9 | 1.6–57.0 | 0.1–5.0 | 1.6–58.0 | 0.1–4.7 |
| TOTAL | 23.0–220.4 | 6.5–61.9 | 0.6–5.5 | 6.6–63.1 | 0.5–5.1 |

***Sensitivity to the methodological approach***

We have seen above that the price and exergy criteria give quite similar allocation of damage, while the approach based on energy burdens more heavily the heat production: the share of total impact goes on from 18.9% to 43.9% (one forth of the total); the choice of one or another methodology changes substantially the calculation of the damage per kWh (see Table 4.20).

**Table 4.20** The different allocations compared

| Criterion | Electricity | Heat | Difference |
|---|---|---|---|
| Energy | 56.1% | 43.9% | 23.5% |
| Exergy | 79.6% | 20.4% | -- |
| Prices | 81.1% | 18.9% | 1.5% |

## 4.5 The Finnish case studies

### 4.5.1 Nature of the case studies

In this section we concentrate solely on the issue of allocating the damage or external costs. The calculation of the overall damages from the two plants considered are discussed in the Finnish contribution to the ExternE national implementation project.

The two cases are selected to represent one relatively simple basic case of coproduction of district heat and electricity, and as the second case a more complicated plant, which supplies in addition to district heat and electricity also steam to a nearby industrial plant. Both cases represent actual plants operational in Finland.

### 4.5.2 Case 1: Coproduction of district heat and electricity

The plant considered is a woodfuel fired relatively small CHP plant producing electricity and heat, which is delivered in the form of hot water into the district heating network of the city of Forssa. The water system is closed and the water returns at a lower temperature to the plant. Production of electricity is calculated as the net production of the plant and is equal to the gross production minus own use at the plant. Production of heat is calculated as the enthalpy difference between the outgoing and return flows of the circulating water. The basic characteristics of the plant are summarised in Table 4.21.

The temperatures of both outgoing and incoming water vary depending on the heating load on the network. The selected values should be near to annual averages weighted based on the heat delivered. The calculations are based on the annual energies, the power levels are given for information only.

The total damages or external costs are calculated on annual basis for the plant. The share to be allocated to the electricity production can be calculated based on several alternative criteria:

**Table 4.21** Technical characteristics of the Forssa district heating CHP

| | Power MW | Annual energy GWh |
|---|---|---|
| Fuel consumption | 72.0 | 259.2 |
| Total delivered energy | 58.9 | 211.8 |
| Electricity (net) | 15.8 | 56.8 |
| District heat (net enthalpy) | 43.1 | 155.0 |
| Temperature of outgoing water ($T_{out}$) | | 100 °C |
| Temperature of return water ($T_{ret}$) | | 50 °C |

***Allocation based on energy content***

The share is based as the share of net electricity production of the total delivered energy:

*Share to be allocated to electricity:* 56.8 GWh/211.8 GWh = 26.8 %.

***Allocation based on exergy content***

The average ambient temperature weighted by heat deliveries is only little above 0 °C. Therefore T0 = 0 °C = 273 K is selected as the reference ambient temperature in the exergy calculations. For hot water the exergy difference of the outgoing and incoming streams can be calculated either using the formula (H = net enthalpy)

$$X = H\,(1 - T_0/(T_{out} - T_{ret})\,\ln(T_{out}/T_{ret}))$$

which is a good approximation, or obtaining the correct values from steam tables or an equivalent computer programme. In the present case the ratio of exergy to enthalpy (X/H) is 0.214 according to the formula and this agrees very well with the steam table values (the difference is of the order of 0.01 % and depends slightly on the pressure).

For electricity exergy is equal to the energy and thus the share of electricity of the total delivered exergy is:

*Share to be allocated to electricity:* 56.8 /(56.8 + 0.214 ´ 155) = 63.1 %.

***Comparison to other approaches***

In the case considered one may argue that the primary product is heat and that at a late stage in the decision making process the two alternatives may have been a heat only plant and a CHP plant. Supposing that this is indeed the case and that the fuel selection and the combustion technologies would have been similar in both cases the increase in plant size and in the total annual fuel consumption are essentially equal to the increase in the total delivered energy. The share to be allocated to electricity would thus be based on energy content. It is, however, questionable whether the actual decision making situation has been as described in this paragraph as there have been many other alternatives available to the decision makers. Furthermore the decision of building a CHP plant affects the extent of the district heating

network. Thus the alternative solution would probably have included more individual heated houses (oil burners, electrical heating, solid fuels).

The typical efficiency of a biomass-fuelled heat only plant of the considered size is about 83 %, while that of a biomass-fuelled electricity only plant is around 35 %. Based on these figures the relative weight of one unit of heat is 42 % of one unit of electricity. This is essentially double to the value based on exergy content and half of that based on energy content.

The relative weights can also be determined based on the monetary value of the products. This is, however, highly ambiguous as the result may depend on the point of measurement. Approximately half of the retail price of electricity is due to the cost of distribution, whereas the cost of distribution is fairly insensitive to the amount of energy transferred and the price is really paid for the basic access to electricity. This fact is reflected in the availability of special tariffs for night-time heating with considerably lower energy component. Similarly for district heating very large part of the cost is due to distribution, but in this case the distribution network is used only to distribute heat, not to provide other basic services.

Furthermore setting of tariffs has been based on various arguments, which include both commercial and political considerations and thus do not necessarily reflect the objective properties of the two forms of energy.

Depending on the approach used in selecting the prices, the price of heat relative to electricity is anywhere between 20 % and 80 %. Thus this approach gives little guidance in judging the relative merits of other approaches.

### 4.5.3 Case 2: Coproduction of district heat, industrial heat and electricity

The plant considered is a peat and biomass fired medium size CHP plant producing electricity and heat, which is delivered in the form of hot water into the district heating network of the city of Jyväskylä and in the form of process steam to a nearby paper mill. The water and steam systems are closed and the water returns at lower temperatures to the plant. Production of electricity is calculated as the net production of the plant and is equal to the gross production minus own use at the plant. Production of heat is calculated as the enthalpy difference between the outgoing and return flows of the circulating water or steam.

The basic characteristics of the plant as used in the calculations are summarised in Table 4.22. These parameters are partly estimated and do not represent accurately the annual averages of the actual plant.

The temperatures of both outgoing and incoming water vary depending on the heating load on the network. The selected values should be near to annual averages weighted based on the heat delivered. The calculations are based on the annual energies, the power levels are given for information only.

**Table 4.22** Technical characteristics of the Rauhalahti district heating and industrial steam CHP

| | Power MW | Annual energy GWh |
|---|---|---|
| Fuel consumption | 340 | 1480 |
| Total delivered energy | 284 | 1213 |
| Electricity (net) | 87 | 350 |
| Industrial steam (enthalpy of steam) | 65 | 300 |
| Industrial heat (net enthalpy) | 57 | 263 |
| District heat (net enthalpy) | 140 | 600 |
| Pressure of industrial steam | | 6 bar |
| Temperature of industrial steam | | 240 °C |
| Pressure of return flow of industrial steam | | 4 bar |
| Temperature of return flow industrial steam | | 70 °C |
| Temperature of outgoing district heating water | | 110 °C |
| Temperature of district heating return water | | 40 °C |

The total damages or external costs are calculated on annual basis for the plant. The share to be allocated to the electricity production can be calculated based on several alternative criteria:

***Allocation based on energy content***

The share is based as the share of net electricity production of the total delivered energy:

*Share to be allocated to electricity:* 350 GWh/1213 GWh = 28.9 %.

***Allocation based on exergy content***

The average ambient temperature weighted by heat deliveries is only little above 0 °C. Therefore $T_0$ = 0 °C = 273 K is selected as the reference ambient temperature in the exergy calculations.

For the hot water supply into the district heating network the exergy is found to be 21.3 % of the enthalpy using either the logarithmic formula given above or steam table values as given by the Exergy calculator. Thus the annual exergy of district heat delivery is 127.7 GWh.

For the industrial steam the exergy can only be calculated from the steam table values. It must also be noted that both the temperature and the pressure of the steam have an essential effect on the ratio of exergy to enthalpy, whereas the pressure of the return water flow has very little effect. The ratio of exergy to enthalpy is 27.0 % for the steam, 12.3 % for the return flow and 28.9 % for their difference or the net flow. The net exergy delivery is thus 76.0 GWh.

For electricity exergy is equal to the energy and thus the share of electricity of the total delivered exergy is:

*Share to be allocated to electricity:* 350/(350 + 128 + 76) = 63.2 %.

***Comparison to other approaches***

For large heat producing plants in Finland CHP is a natural and automatic choice. Thus it is not possible to divide the decision making process in successive independent steps. The design of the plant is based on a simultaneous consideration of the three main loads (hot water, steam and electricity). Thus estimating the share of any of the loads through subtraction leads to great ambiguities.

The typical efficiency of a peat or biomass-fuelled heat only plant of the considered size is about 86 %, while that of a biomass-fuelled electricity only plant is around 38 %. Based on these figures the relative weight of one unit of heat is 44 % of one unit of electricity. This is essentially double to the value based on exergy content and half of that based on energy content.

Basing the allocation of damages on the relative prices of the products is even more impractical in this case than in the previous case, because the price of the industrial steam is not available. Even if it were, the difficulties related to the costs of transportation and distribution are also even more difficult to solve than in the case were only district heat and electricity were compared.

## 4.6 Conclusions

In the above sections we have developed a consistent approach to integrate combined heat and power producing technologies into the existing ExternE accounting framework. While the priority impact pathways are similar to other fuel cycles analysed before, we have very much focused on the allocation of external costs from a cogeneration plant between electricity and heat.

Although the allocation based on operational characteristics, including e.g. credits for heat or electricity that might be considered as by-products in specific cases, is commonly used for the calculation of 'internal' costs, we conclude that the allocation based on thermodynamic parameters, namely the exergy content of the product streams, is the most appropriate approach within the ExternE context. Thermodynamic parameters are clearly defined by the power plant's characteristics and do not depend on e.g. the technical parameters of technologies that might be substituted by the CHP plant. In theory, the price of the final products is probably the best indicator for the 'utility' of the products and thus an appropriate indicator for allocation, but in practice tariffs might be disturbed because of various commercial or political reasons and thus do not necessarily reflect the objective properties of the two forms of energy. However, results from the German and the Italian case studies show that the allocation based on the exergy and the price indicators lead to similar results.

The methodology developed has been applied to a wide range of CHP technologies, including different fuels (natural gas, coal, biomass) and different locations (Germany, Italy, Finland), thus reflecting the different conditions under which a CHP plant might be operated. In the

case of the Finish CHP plant providing hot water for a district heating network of a city, one may argue that the primary product is heat, while in the Italian case the CHP plant is clearly operated towards electricity generation, leading to a different share of external costs allocated to either heat or electricity.

The current results provide a framework to calculate external costs from a CHP plant per unit electricity or heat production consistent with the overall ExternE approach. However, we want to emphasise that a direct comparison between a CHP plant and a plant producing heat or electricity only remains difficult, because we can always substitute alternative technologies. To compare the overall external costs resulting from a supply system with and without cogeneration we always have to take into account the characteristics of the respective full energy system.

## 4.7 References

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# 5. ANALYSIS OF NEW TECHNOLOGIES

## 5.1 IGCC technology

### 5.1.1 Reference technology

The technology chosen for this study has been Integrated Gasification Combined Cycle (IGCC). A power plant using this technology has been built in Puertollano, 230 km south from Madrid (Spain), by Elcogas, a consortium formed by several European utilities and engineering companies. This technology is one of today's most promising electricity generation options, both from the environmental and the efficiency point of view. It will allow an efficient and environmentally-friendly use of national coal, and also of a refinery residue, petroleum coke. When operating, this plant will be the world's largest IGCC plant.

The power plant will have a total gross output of 335 MW, and will be fueled by a mixture of coal from local coalmines and petroleum coke from a refinery close to the IGCC plant. Natural gas will also be used as a backup fuel.

The major problem for the assessment of this technology is the definition of the boundaries of the fuel cycle, due to the use of petroleum coke. If we consider petroleum coke a residue of the refinery, then we may exclude from the fuel cycle the refinery process, and its upstream stages, based on the marginal analysis. However, petroleum coke is a traded product, with several alternative uses, and so it should not be considered a residue, but a by-product from the refinery. Therefore, the fuel cycle should include the refining and upstream stages. The effects of this stages will be attributed to the coke production based on its share of the total refinery products (see section 5.1.2). Nevertheless, the construction and dismantling stages of the refinery will not be taken into account, since they will not produce marginal impacts, as it is hardly conceivable to build a new refinery just for producing coke.

A diagram of the fuel cycle considered is shown in Figure 5.1. Each of these stages will be described in the following sections.

***Oil extraction***

Oil refined in Spain comes mainly from the Persian Gulf, Mexico, Nigeria, Libya and Russia. However, we may assume that the extraction conditions for these countries are similar. The oil extraction process produces methane emissions from the wells, and also atmospheric emissions from related activities.

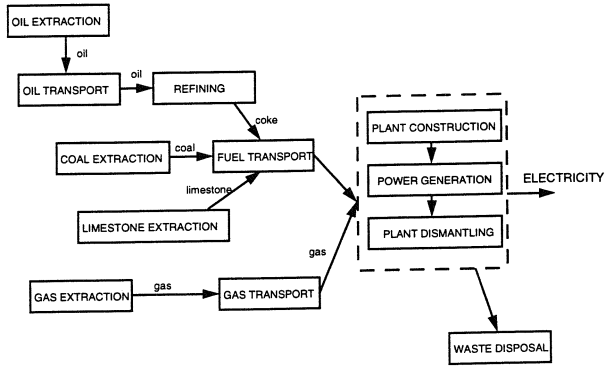



**Figure 5.1** The IGCC fuel cycle

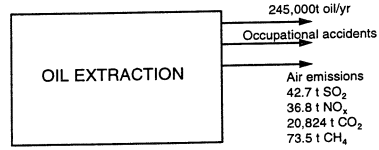



**Figure 5.2** Oil extraction.

The construction and dismantling of the oil wells, as well as labour needed for this stage, have not been considered, as their impacts are expected to be negligible.

For our calculations of the emissions, we have based on the data provided by IEFE (1995) for the ITGCC fuel cycle, since they are expressed for OPEC oil extraction, which is the one to be applied also for our case. Their data have been corrected with the crude oil requirement for this case, which is 0.126 Mt/TWh. The resulting emissions are shown in the following figure.

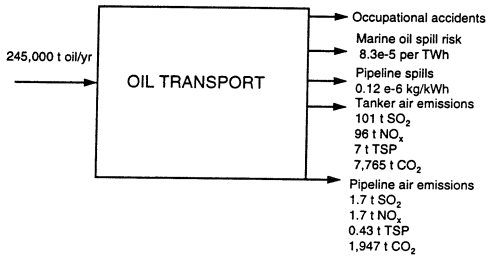



**Figure 5.3** Oil transport

***Oil transport***

Oil arrives at the refinery by an oil pipeline, once brought from the countries mentioned above by tankers. We may assume that the transport conditions are similar to those of Italian oil, so we will use the data provided by IEFE (1995) for the calculation of the atmospheric emissions of the oil transport, and also for the possible impact of oil spills.

Data on oil pipeline-related emissions and oil spills have been taken from the ExternE report (EC, 1995), assuming that the transport distance will be similar. All data have been corrected for the crude oil requirements of the IGCC fuel cycle, which is 0.126 Mt/TWh.

***Oil refining and coke production***

Oil will be refined in Puertollano refinery, sited some 6 km away from the IGCC power plant.

The major impacts of this stage are caused by the air emissions of the refinery. However, not all the refining processes have to be considered, only those involved in petroleum coke production, namely topping (atmospheric distillation), vacuum distillation, and coking.

Unfortunately, the obtention of emission rates for the different plants of the refinery has been impossible. Therefore, the total emissions of the refinery (obtained from CORINAIR) have been assigned to the different stages following the same distribution as for the Tirrenic refinery described by IEFE (1995). We have assumed that the coking unit emissions are similar to those of the visbreaking one.

The share of emissions attributable to the IGCC fuel cycle has been determined based on the mass relationship between coke and the input for each stage. The crude oil input and coke

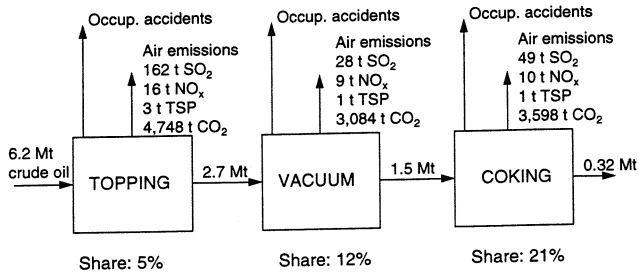



**Figure 5.4** Oil refining for coke production

output are real figures for the Puertollano refinery. The percentage of feedstock going into the next stage has been taken from the ITGCC fuel cycle study (IEFE, 1995).

It will be assumed that all the emissions of the refinery stage will come through a stack with 100 m height and 1 m diameter. The flue gas volume has been estimated in 19,660 $Nm^3/h$, based on the calculation procedure proposed by IEFE (1995). Flue gas temperature is 473K. The characteristics of the coke produced are shown in Table 5.1.

***Coal extraction***

Coal is extracted from the Emma open cast mine, sited some 20 km away from the power plant. Its estimated coal reserves are 60 Mt, of which some 320 kt will be used yearly by the IGCC power plant. This represents about one third of the total coalmine production. Coal characteristics are shown in Table 5.2.

To extract the coal, a first drilling and blasting is done, in order to separate the topsoil, and this topsoil is spreaded over the mine refuse at the back of the exploitation area. Coal is extracted with diggers and transported to the treatment area, where it is crushed, screened, and processed in a sink and float plant.

Atmospheric emissions are produced during this stage due to the fugitive dust emissions from the extraction process, to the dust emissions associated to the coal treatment, and to the vehicle emissions.

**Table 5.1** Characteristics of the petroleum coke

| PROXIMATE ANALYSIS | % weight |
|---|---|
| Moisture | 7.00 |
| Ash | 0.26 |
| Volatiles | 13.00 |
| Fixed carbon | 79.74 |
| ULTIMATE ANALYSIS | % weight |
| Moisture | 7.00 |
| Ash | 0.26 |
| Carbon | 82.21 |
| Hydrogen | 3.11 |
| Nitrogen | 1.90 |
| Oxygen | 0.02 |
| Sulfur | 5.50 |
| Chlorine | - |
| Phosphorus | - |
| GROSS CALORIFIC VALUE | 31.99 MJ/kg |
| NET CALORIFIC VALUE | 32.65 MJ/kg |

**Table 5.2** Characteristics of the coal from the Emma mine

| PROXIMATE ANALYSIS | % weight |
|---|---|
| Moisture | 11.8 |
| Ash | 41.10 |
| Volatiles | 18.83 |
| Fixed carbon | 28.27 |
| ULTIMATE ANALYSIS | % weight |
| Moisture | 11.8 |
| Ash | 41.10 |
| Carbon | 36.22 |
| Hydrogen | 2.48 |
| Nitrogen | 0.81 |
| Oxygen | 662 |
| Sulfur | 0.93 |
| Chlorine | 0.04 |
| Phosphorus | 0.01 |
| GROSS CALORIFIC VALUE | 13.58 MJ/kg |
| NET CALORIFIC VALUE | 13.10 MJ/kg |

In order to avoid the physical impact of the mine works, soil restoration is permanently carried out at the back of the exploitation area.

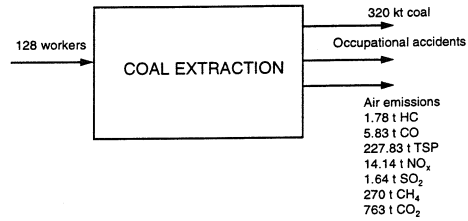



**Figure 5.5** Coal extraction

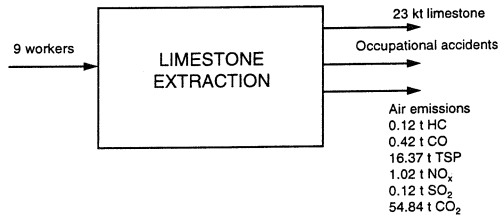



**Figure 5.6** Limestone extraction

It has to be noted that, although TSP emissions are quite high, they are mostly large particles, which do not travel long distances, staying for the most part within the mine limits.

***Limestone extraction***

Limestone is extracted from a limestone quarry 30 km away from the plant. The yearly requirements for the power plant are around 23 kt. For the estimation of the labour needed for its extraction, the same productivity as for open cast mining has been used.

**Table 5.3** Average composition of Hassi R'Mel natural gas

| Components | Symbol | Normal composition (%v) |
|---|---|---|
| Methane | $CH_4$ | 91.2 |
| Ethane | $C_2H_6$ | 7.4 |
| Propane | $C_3H_8$ | 0.8 |
| Butane | $C_4H_{10}$ | 0.1 |
| Nitrogen | $N_2$ | 0.5 |
| Carbon dioxide | $CO_2$ | - |
| Hydrogen sulphide | $H_2S$ | - |
| Others | | - |

Source: SEDIGAS (1993)

***Gas extraction and transport***

Gas will come from the Algerian gas field of Hassi R'Mel. Daily production in this gas field is $3.7 \cdot 10^7$ $m^3$, while the amount needed by the power plant is $96 \cdot 10^3$ $m^3$, that is, a negligible amount. The average composition of the gas is shown in Table 5.3.

The gas will be transported to the power plant by the Maghreb-Europe pipeline, with a total length of 1,500 km. The pipeline is buried, and avoids large population centres and ecologically sensitive areas. Its environmental impact is expected to be very small, since the land affected is quite reduced, and it is restored in a few months.

Valves are installed every 20 km along the pipeline, with associated equipment on the ground. Two compressor stations have been built in the section from the gas field to the power plant. However, only one of them is needed for the current gas volume transported. These compressor stations have an installed power of 25 MW, with an average consumption of 3,500 $m^3/h$ of the natural gas.

The personnel needed for operation and maintenance of the pipeline is around 260 people, most of them for the African section.

Due to the buried installation of the pipeline, its dismantling will not be considered.

A diagram with the inputs and outputs of this stage is shown in the following figure.

Of the burdens identified, we will only assign to the fuel cycle studied the share corresponding to the power plant consumption of the total gas volume transported.

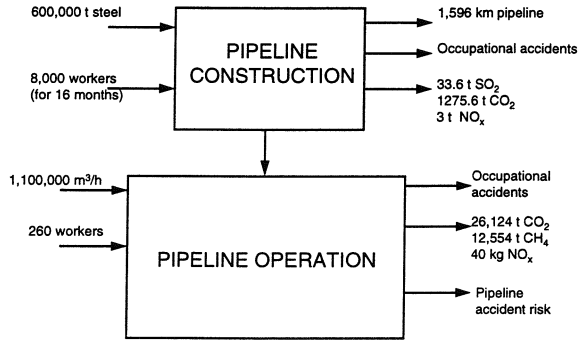



**Figure 5.7** Gas transport

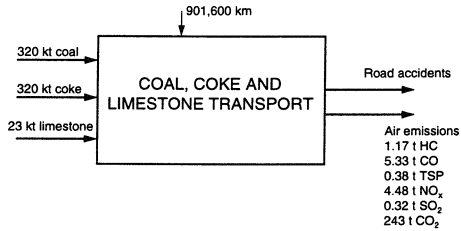



**Figure 5.8** Fuel transport

***Fuel transport***

Of the three types of fuel required by the power plant, gas transport has already been addressed. The rest of the fuel, and the limestone, will be transported by heavy-duty, diesel-powered truck. Average transport distances are 20 km for the coal, 6 km for the coke, and 30 km for the limestone. Because of these short distances, road damages will not be taken into account, only atmospheric emissions and road accidents will be considered.

***Power generation***

The three main units of the IGCC power plant are the gasifier, the air separation unit (ASU), and the combined cycle (CC). Other basic units of the plant are the coal gas treatment plant, the fuel preparation system, and the cooling water system.

The coal, coke and limestone are piled in stacks, so that a minimum fuel stock for 45 days is assured. Then they are transported to the fuel preparation unit, where they are crushed, sieved and dried. These operations are carried out in a nearly pure nitrogen atmosphere, to avoid the fire risk due to the fine pulverization of the solid fuel.

The gasifier consumes 110 t/h of fuel (50% coal and 50% coke) and 4 - 7 t/h of limestone. The homogenized fuel is fed to the gasifier, that consists of a reaction chamber inside of a pressurised vessel. The high reaction temperature permits the liquefaction of the coal ashes which slide down the walls of the chamber. The slag is solidified with water and disposed without loosing pressure inside of the gasifier.

The product gas exits the upper part of the gasifier and it is cooled down generating steam. Flying ashes from the product gas are eliminated by ceramic candle filters and recirculated to the gasifier. The product gas is cleaned up of acid gases in a two stage process. In the first stage, acid gases and ammonia are retained or reformed. Hydrogen sulphide that remains in the flue gas is extracted in a second stage with organic solvents and transformed in elemental sulphur in a Claus unit.

The clean gas is fed as fuel of the CC. However, it will be possible to fuel the CC unit with natural gas. This situation will happen the first year of operation of the plant, during the gasifier scheduled stops, and in emergency situations when the plant needs an inmediate start-up.

The gas combustion system is of multichamber type and has been specially designed to allow minimum generation levels of nitrogen oxides during combustion. The gas turbine velocity is 3,000 r.p.m. and the power generated is 182.5 MW. The heat recovery boiler for the exhaust gases of the gas turbine works in three pressure levels. The boiler's structure supports the stack, which is at the top of the boiler.

The steam generated in the heat recovery boiler feeds a steam turbine with a power of 137.5 MW.

Altogether, the net power output is 300 MW (based on ISO guidelines). The annual working hours will be 6,500.

The air separation unit produces oxygen for the gasification and nitrogen for inerting in the fuel preparation unit and for diluting the coal gas that enters the gas turbine.

The cooling water system consists of a cooling tower of 120 m height and 82 m diameter, and the auxiliary pumps needed to recirculate the water. The water supply required is around 600 $m^3/h$, which will come from reservoirs in the area.

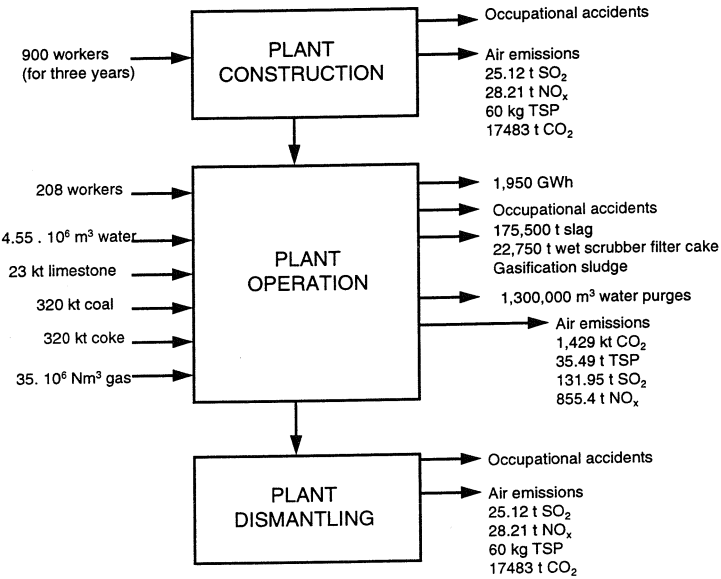



**Figure 5.9** Power generation

Flue gas is emitted through a stack of 60 m height, and 7 m diameter. The flue gas volume is around 1.4 e6 $Nm^3/h$, with a temperature around 383°K.

The net plant efficiency will be around 45%. Its technical lifetime will be of 25 years, and the workforce required for operation and maintenance is 208 workers.

The plant construction period will last 3 years, employing some 900 workers. The atmospheric emissions during this period have been obtained from the SAFIRE dataset of the DGXII.

All the process, with its material requirements, is shown in the following figure.

***Waste disposal***

The two major characteristic waste products of the IGCC plant are fly ashes and vitrified slag from the gasifier, and filter cake from the effluent of the gasifier gas wet scrubbing.

Slag has been shown to be not toxic, so it will be disposed in the coal mine together with the mine refuse.

**Table 5.4** Matrix of activities-impacts of the fuel cycle

| Stage - Activity | Human health | | | Ecosystems | | | | Other | |
|---|---|---|---|---|---|---|---|---|---|
| | Public Health | Occupational Health | Accidents | Crops | Forests | Ecosystems | Global Warming | Materials | Visual intrusion |
| *Mining* | | | | | | | | | |
| Coal mining | √ | √ | √ | √ | √ | √ | √ | √ | √ |
| Coal treatment | √ | √ | √ | √ | √ | √ | √ | √ | √ |
| Oil extraction | √ | √ | √ | √ | √ | √ | √ | √ | √ |
| Limestone extration | √ | √ | √ | √ | √ | √ | √ | √ | √ |
| *Oil refining* | √ | √ | √ | √ | √ | √ | √ | √ | √ |
| *Transport* | | | | | | | | | |
| Coal transport | √ | √ | √ | √ | √ | √ | √ | √ | |
| Coke transport | √ | √ | √ | √ | √ | √ | √ | √ | |
| Gas transport | | | √ | | | | √ | | |
| Limestone transport | √ | √ | √ | √ | √ | √ | √ | √ | |
| *Generation* | | | | | | | | | |
| Plant construction and dismantling | √ | √ | √ | √ | √ | √ | √ | √ | √ |
| Electricity generation | √ | √ | √ | √ | √ | √ | √ | √ | √ |

***Identification and prioritization of the impacts***

A summary of the impacts produced by the activities described above is shown in Table 5.4. These impacts have been prioritized, as shown in Table 5.5.

The list of priority impacts for the fuel cycle going to be assessed is given below:

- Effects on public health by $PM_{10}$, $SO_2$ and aerosols,
- Effects on global warming by $CO_2$ and $CH_4$,
- Effects on occupational health by accidents,
- Effects on materials by $SO_2$, $NO_x$, ozone and acid rain,
- Effects on ecosystems by ozone and acid rain,
- Effects on crops by ozone and $SO_2$.

However, since no ozone dispersion model is available, the effects of ozone will not be assessed.

**Table 5.5** Priority impacts

| Burden | Receptor | Impact | Priority |
|---|---|---|---|
| Primary atmospheric emissions: | | | |
| $PM_{10}$ | Human | Public health | High |
| $SO_2$ | Human | Public health | Medium |
| | Crops | Loss of yields | Low |
| | Materials | Degradation | High |
| $CO_2$ | Ecosystems | Global Warming | High |
| $CH_4$ | Ecosystems | Global Warming | Medium |
| Secondary atmospheric emissions: | | | |
| Aerosols | Human | Public health | High |
| Ozone | Human | Public health | High/nq |
| | Crops | Loss of yields | High/nq |
| | Ecosystems | Degradation | High/nq |
| | Materials | Corrosion | High/nq |
| Acid rain | Ecosystems | Degradation | High |
| | Materials | Degradation | High |
| Solid residues: | | | |
| Ashes and slag | Ecosystems | Soils lixiviation | Medium/nq |
| Water sludge | Ecosystems | Soils lixiviation | Medium/nq |
| Mine refuse | Ecosystems | Soils lixiviation | Low |
| Liquid effluents: | Ecosystems | Degradation | Medium/nq |
| | Public | Public health | nq |
| Other burdens: | | | |
| Noise | Workers | Occupational health | Low/nq |
| | Public | Nuisance | Low/nq |
| Accidents | Workers | Occupational health | High |
| | Public | Public health | Low/nq |
| Road use | Public | Physical damage | Low |
| Visual intrusion | Public | Amenity | Low |

nq: not quantified

### 5.1.2 Methodological issues

No new major methodological issues have been identified for this technology. The assessment of the IGCC technology is basically similar to conventional combustion technologies, except that atmospheric pollutant emissions are reduced, so the impact pathways to be analyzed are the same as for these conventional technologies.

Only two controversial issues have been identified: the definition of the fuel cycle, and the air emissions from the mining stage.

The definition of the fuel cycle is more complex than for other cases, due to the number of fuels involved, and their origin. This is specially important for petroleum coke, which comes from oil refining.

Generally, coke is considered a residue of the refining process. Therefore, if we follow the marginal approach proposed by the ExternE methodology, we should not take into account the impacts produced by the refining stage, since these would be produced even if the coke is not used.

However, petroleum coke has a trading price, and several alternative uses in the market, so it should not be considered a residue, but rather a by-product of the refinery. Moreover, the amount of petroleum coke can be changed modifying the operation parameters of the refinery. So we should conclude that the production of coke is really affecting the impacts of the refining stage, and so they should be taken into account (as well as the upstream impacts of the refining process).

Here another problem arises, which is the attribution of the refinery impacts to the petroleum coke. This has been done following the approach proposed by IEFE (1995), that is, considering those refinery processes in which coke is involved. For these processes, burdens should be attributed based on the energy content of the feedstock input related to that of the final coke output. The problem here is that very detailed information on energy contents, and atmospheric emissions, is required, which has been impossible to obtain. Therefore, the share has been estimated based on mass, rather than on energy, and the emissions for each stage have been scaled from an Italian refinery described by IEFE (1995), based on the actual Puertollano refinery emissions obtained from CORINAIR.

Regarding the air emissions from the mining stage, it has to be noted that TSP emissions from the mining make up for 80% of the total TSP emissions of the fuel cycle. However, it has to be reminded that it is not TSP which produce the significant health damages, but $PM_{10}$, that is, the TSP fraction under 10 μm size. And most of the TSP emitted during the mining are larger than this size, so, firstly, they would not be transported over long distances, and secondly, they would not cause significant health damages. On the other hand, TSP emissions from the power generation stage are mostly $PM_{10}$, so, if we actually consider $PM_{10}$ instead of TSP, then it is this stage which emits the most part of this pollutant. Therefore, it is considered that the incorporation of coal mining TSP emissions as if they were coming from power generation would produce a very large overestimation of health damages, and so it has been decided not to include them.

### 5.1.3 Quantification of results

For the quantification of the impacts, not all of them have been considered. Effects of $PM_{10}$, $SO_2$, $NO_x$, and aerosols will only be assessed for the oil refining and power generation stages, since these are the major polluting stages. Although the emissions from oil transport by tanker are also significant, the modelling of their dispersion is quite complex, and thus it has not been attempted.

**Table 5.6** Impacts and damages of the oil extraction stage

| Impact | Burden | Impacts | | | Damages | | |
|---|---|---|---|---|---|---|---|
| | | Unit | per TWh | $\sigma_g$ | mECU/kWh | ECU/t poll. | $\sigma_g$ |
| **Occupational accidents** | | | | | | | |
| Deaths | | | 4.7e-3 | | 1.2e-2 | na | A |
| Major injuries | | | 5.5e-3 | | 6.8e-4 | na | A |
| Minor injuries | | | 4.9e-2 | | 9.4e-5 | na | A |
| **Subtotal** | | | | | **1.3 e-2** | | |
| **Global warming** | $CO_2$ | | | | 4.1e-2-1.48 | 3.8-139 | C |
| | | | | | 1.9e-1-4.9e-1 | 18-46 | C |
| | $CH_4$ | | | | 6.0e-3 | 159 | C |

na: not applicable

**Table 5.7** Impacts and damages of the oil transport stage.

| Impact | Burden | Impacts | | | Damages | | |
|---|---|---|---|---|---|---|---|
| | | Unit | per TWh | $\sigma_g$ | mECU/kWh | ECU/t poll. | $\sigma_g$ |
| **Occupational accidents** | | | | | | | |
| Deaths | | | 2.0e-2 | | 5.2e-2 | na | |
| Major injuries | | | 1.2e-1 | | 1.5e-2 | na | |
| Minor injuries | | | 1.6 | | 3.1e-3 | na | |
| **Marine oil spill** | | | 8.3e-5 | | 7.7e-2 | | |
| **Pipeline oil spill** | | | 1.2e-4 | | nq | | |
| **Subtotal** | | | | | **1.5 e-1** | | |
| **Global warming** | $CO_2$ | | | | 1.9e-2-6.9e-1 | 3.8-139 | |
| | | | | | 9e-2-2.3e-1 | 18-46 | |

na: not applicable

On the other hand, since the impact of $CO_2$ and $CH_4$ emissions is global, these will be assessed for all the stages of the fuel cycle.

***Oil extraction***

The major impact of this stage is the global warming produced by the GHG emissions produced by flaring and venting, and by the associated processes. Occupational accidents have been calculated based on the accident rates provided by the ExternE report (EC, 1995).

***Oil transport***

During this stage, the major impact is also caused by atmospheric emissions from the tanker and the pipeline. However, only the impact of $CO_2$ emissions has been assessed, since the dispersion modelling of the rest of the pollutants is much more complex, specially taking into account that they are not emitted at a point source, but along a variable trajectory. The effect of pipeline oil spills has also not been quantified, since the amount of oil spilt is very small. Occupational accidents estimation has also been based on the EXTERNE report (EC, 1995), as well as marine oil spill risk.

***Oil refining and coke production***

$SO_2$ emissions from the refinery cause the most significant impacts of this stage. These emissions are rather high, the highest of the fuel cycle, due to the high sulphur content of the oil used for coke production.

Occupational accidents have been again estimated using the accident rates proposed by the ExternE report (EC, 1995)

***Coal extraction***

The most relevant impacts from coal extraction come from the occupational accidents, due to the high accident rates of mining. GHG emissions also cause significant impacts within this stage.

***Limestone extraction***

The impacts from this stage are very small.

***Gas extraction and transport***

Most of the impacts from this stage come from the gas leaks of the pipeline. Both the construction and operation of the pipeline are included, although the gas extraction stage has not been included, due to the lack of data. However, it may be seen that the impacts from this stage are not really significant, as the amount of gas used by the power plant is small.

**Table 5.7** Impacts and damages of the oil refining stage

| Impact | Burden | Impacts | | $\sigma_g$ | Damages | | $\sigma_g$ |
|---|---|---|---|---|---|---|---|
| | | Unit | per TWh | | mECU/kWh | ECU/t poll. | |
| **Human health** | | | | | | | |
| Chronic YOLL | TSP | years | 1.82e-1 | | 1.53e-2 | 5967 | B |
| | Nitrates | years | 1.25 | | 1.06e-1 | 5906 | B? |
| | Sulfates | years | 8.00 | | 6.74e-1 | 5499 | B |
| Acute YOLL | $SO_2$ | years | 1.05e-1 | | 1.62e-2 | 132 | B |
| Morbidity | Nitrates | cases | 7.81e+1 | | 1.36e-2 | 758 | A-B? |
| | $SO_2$ | cases | 3.98e-2 | | 3.14e-4 | 3 | A-B |
| | Sulfates | cases | 4.99e+2 | | 8.46e-2 | 690 | A-B |
| | TSP | cases | 1.13e+1 | | 1.97e-3 | 768 | A-B |
| **Crops** | $SO_2$ | dt yield loss | 286 | | 1.86e-3 | 15 | |
| | N dep. | kg fertilizer added | 730 | | -1.25e-5 | -1 | |
| | Ac. dep. | kg lime added | 6.07e4 | | 1.04e-3 | na | |
| **Ecosystems** | N dep. | $km^2$ exceed. area | 0 | | 0 | 0 | |
| | $SO_2$ | $km^2$ exceed. area | 0 | | 0 | 0 | |
| | $NO_x$ | $km^2$ exceed. Area | 0 | | 0 | 0 | |
| **Materials** | $SO_2$ | $m^2$ maint. area | 3.92e+2 | | 6.68e-3 | 55 | |
| **Occupational accidents** | | | | | | | |
| Deaths | | | 1.3e-3 | | 3.4e-3 | na | |
| Major injuries | | | 3.5e-2 | | 4.3e-3 | na | |
| Minor injuries | | | 1.3 | | 2.5e-3 | na | |
| **Ozone** | $NO_2$ | | | | 2.7e-2 | 1500 | |
| **Subtotal** | | | | | **9.6e-1** | | |
| **Global warming** | $CO_2$ | | | | 2.2e-2-8.2e-1 | 3.8-139 | |
| | | | | | 1.1e-1-2.7e-1 | 18-46 | |

na: not applicable

**Table 5.8** Impacts and damages of coal extraction

| Impact | Burden | Impacts | | | Damages | | |
|---|---|---|---|---|---|---|---|
| | | Unit | per TWh | $\sigma_g$ | mECU/kWh | ECU/t poll. | $\sigma_g$ |
| **Occupational accidents** | | | | | | | |
| Deaths | | | 4.7e-2 | | 1.2e-1 | na | |
| Major injuries | | | 7.6e-2 | | 9.4e-3 | na | |
| Minor injuries | | | 24.1 | | 4.6e-2 | na | |
| **Subtotal** | | | | | **1.76e-1** | | |
| **Global warming** | $CO_2$ | | | | 1.49e-3-5.4e-2 | 3.8-139 | |
| | | | | | 7.04e-3-1.8e-2 | 18-46 | |
| | $CH_4$ | | | | 2.2e-2 | 159 | |

na: not applicable

**Table 5.9** Impacts and damages of limestone extraction

| Impact | Burden | Impacts | | | Damages | | |
|---|---|---|---|---|---|---|---|
| | | Unit | per TWh | $\sigma_g$ | mECU/kWh | ECU/t poll. | $\sigma_g$ |
| **Occupational accidents** | | | | | | | |
| Deaths | | | 2.3e-3 | | 6.0e-3 | na | |
| Major injuries | | | 1.0e-2 | | 1.2e-3 | na | |
| Minor injuries | | | 0.53 | | 1.0e-3 | na | |
| **Subtotal** | | | | | **8.2e-3** | | |
| **Global warming** | $CO_2$ | | | | 1.07e-4-3.9e-3 | 3.8-139 | |
| | | | | | 5.06e-4-1.29e-3 | 18-46 | |

na: not applicable

**Table 5.10** Impacts and damages of gas extraction and transport

| Impact | Bur-den | Impacts | | | Damages | | |
|---|---|---|---|---|---|---|---|
| | | Unit | per TWh | $\sigma_g$ | mECU/kWh | ECU/t poll. | $\sigma_g$ |
| **Occupational accidents- EU** | | | | | | | |
| Deaths | | | 2.8e-5 | | 7.3e-5 | na | |
| Injuries | | | 1.3e-2 | | 2.4e-4 | na | |
| **Occupational accidents- Non-EU** | | | | | | | |
| Deaths | | | 1.1e-4 | | 2.9e-4 | na | |
| Injuries | | | 5.8e-2 | | 1.1e-3 | na | |
| **Accident risk- EU** | | | | | | | |
| Deaths | | | 3.1e-6 | | 8.2e-6 | na | |
| Injuries | | | 1.2e-5 | | 2.2e-7 | na | |
| **Accident risk- Non-EU** | | | | | | | |
| Deaths | | | 3.3e-6 | | 8.5e-6 | na | |
| Injuries | | | 1.3e-5 | | 2.3e-7 | na | |
| **Subtotal** | | | | | **1.72e-3** | | |
| **Global warming** | $CO_2$ | | | | 2.5e-4-9.2e-3<br>1.19e-3-<br>3.05e-3 | 3.8-139<br>18-46 | |
| | $CH_4$ | | | | 5.0e-3 | 159 | |

na: not applicable

**Table 5.11** Impacts and damages of fuel transport

| Impact | Bur-den | Impacts | | | Damages | | |
|---|---|---|---|---|---|---|---|
| | | Unit | per TWh | $\sigma_g$ | mECU/kWh | ECU/t poll. | $\sigma_g$ |
| **Road accidents** | | | | | | | |
| Deaths | | | 3.8e-3 | | 9.9e-3 | na | |
| Major injuries | | | 1.0e-2 | | 1.2e-3 | na | |
| Minor injuries | | | 1.9e-2 | | 3.6e-5 | na | |
| | | | | | **1.1e-2** | | |
| **Global warming** | $CO_2$ | | | | 4.74e-4-<br>1.73e-3<br>2.24e-3-<br>5.73e-3 | 3.8-139<br><br>18-46 | |

na: not applicable

***Fuel transport***

As said before, we have not considered road damages, due to the very small distance to be travelled for fuel transport. Road accidents have been considered on the whole distance.

As for the atmospheric emissions of the vehicles, its impact has been considered negligible, due to its very small contribution to the total fuel cycle emissions. Only greenhouse gas emissions have been taken into account, due to their global nature.

***Power generation***

This stage includes the impacts of the plant construction and decomissioning, although they are negligible except for occupational accidents. The major impact of this stage is that caused by $CO_2$ and $NO_x$ emissions, being the rest quite small because of the gasification technology. Of the impacts quantified, the most important ones are chronic mortality and global warming.

***Waste disposal***

As mentioned before, waste products are considered non-toxic, and are disposed in the coal mine, together with the mine refuse. Therefore, it is expected that the impact of these wastes will be negligible.

***Summary***

The damage costs of all process steps of the reference energy system are put together in Table 5.13.

### 5.1.4 Conclusions

Total impacts from this fuel cycle are quite small due to the characteristics of the generation technology. Major impacts come from the power generation stage, where impacts of $CO_2$ emissions on global warming and $NO_x$ emissions on human health are the most important ones. Within the upstream stages of the fuel cycle, the oil refining and coke production stage deserves special attention. This stage originates the highest $SO_2$ emissions of the whole fuel cycle, due to the high sulfur content of the oil used for coke production, and these emissions produce a significant impact on human health.

**Table 5.12** Impacts and damages of power generation

| Impact | Bur-den | Impacts | | $\sigma_g$ | Damages | | $\sigma_g$ |
|---|---|---|---|---|---|---|---|
| | | Unit | per TWh | | mECU/kWh | ECU/t poll. | |
| **Human health** | | | | | | | |
| Chronic YOLL | TSP | years | 1.17 | | 9.88e-2 | 5429 | B |
| | Ni-trates | years | 3.58e+1 | | 3.02 | 6884 | B? |
| | Sul-fates | years | 3.82 | | 3.22e-1 | 4759 | B |
| Acute YOLL | $SO_2$ | years | 4.90e-2 | | 7.50e-3 | 111 | B |
| Morbidity | Ni-trates | cases | 2.23e+3 | | 3.88e-1 | 884 | A-B? |
| | $SO_2$ | cases | 1.87e-2 | | 1.48e-4 | 2 | A-B |
| | Sul-fates | cases | 2.38e+2 | | 4.01e-2 | 593 | A-B |
| | TSP | cases | 7.30e+1 | | 1.27e-2 | 698 | A-B |
| **Crops** | $SO_2$ | dt yield loss | 1.35e+2 | | 9.96e-4 | 15 | |
| | N dep. | kg fertilizer added | -3.45e+4 | | -5.91e-4 | -1 | |
| | Ac. dep. | kg lime added | 1.57e+5 | | 2.69e-3 | na | |
| **Ecosystems** | N dep. | $km^2$ ex-ceed. area | 0 | | 0 | 0 | |
| | $SO_2$ | $km^2$ ex-ceed. area | 0 | | 0 | 0 | |
| | $NO_x$ | $km^2$ ex-ceed. area | 0 | | 0 | 0 | |
| **Materials** | $SO_2$ | $m^2$ maint. area | 1.17e+3 | | 1.79e-2 | 265 | |
| **Occupational accidents** | | | | | | | |
| Deaths | | | 3.3e-2 | | 8.5e-2 | na | |
| Major injuries | | | 1.9e-1 | | 2.4e-2 | na | |
| Minor injuries | | | 7.8 | | 1.5e-2 | na | |
| **Ozone** | $NO_2$ | | | | 6.8e-1 | 1500 | |
| **Subtotal** | | | | | **4.72** | | |
| | | | | | 2.79-101.99 | 3.8-139 | |
| **Global warming** | $CO_2$ | | | | 13.21-33.75 | 18-46 | |

na: not applicable

**Table 5.13** Summary of the quantified damages

| | mECU/kWh | $\sigma_g$ |
|---|---|---|
| **POWER GENERATION** | | |
| Public health | | |
| Mortality*- YOLL *(VSL)* | 3.45 | B |
| *of which TSP* | *0.1* | |
| *$SO_2$* | *0.33* | |
| *$NO_x$* | *3.02* | |
| Morbidity | 0.44 | |
| Accidents | nq | A |
| Occupational health | 0.124 | A |
| Major accidents | nq | |
| Crops | 3.1e-3 | B |
| *of which $SO_2$* | *9.96e-4* | |
| Ecosystems | ng | B |
| Materials | 1.79e-2 | B |
| Ozone | 0.68 | |
| Global warming | | C |
| Conservative 95% confidence interval | 2.79-101.99 | |
| Illustrative restricted range | 13.21-33.75 | |
| **OTHER FUEL CYCLE STAGES** | | |
| Public health | 0.912 | A |
| Occupational health | 0.29 | A |
| Crops | 2.89e-3 | B |
| Materials | 6.68e-3 | A |
| Global warming | | C |
| CO2 Conservative 95% confidence interval | 0.084-3.06 | |
| Illustrative restricted range | 0.4-1.02 | |
| CH4 | 0.033 | |
| **SUBTOTAL** | | |
| Non-global warming impacts | **6.04** | |
| Global warming | | |
| Conservative 95% confidence interval | **2.9-105.1** | |
| Illustrative restricted range | **13.64-34.8** | |

### 5.1.5 References

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## 5.2 Fluidised Bed Combustion

### 5.2.1 Introduction

Applying new thermodynamic cycles is one option to improve the efficiency and to decrease the environmental impact of electricity production from fossil fuels. One of the new thermodynamic cycles of interest is fluidised bed combustion (FBC). FBC was first seriously considered in the early 1960s, as an alternative to conventional pulverised fuel firing systems. Due to environmental considerations and due to the fuel flexibility of FBC the interest has further increased. In the following years research on this technology was promoted and at the beginning of the nineties several hundreds facilities had been built. At the present time, fluidised bed boilers are being intensively developed in Sweden, the UK, France, Finland, Germany, and the USA. Although units up to 500 $MW_{th}$ are currently in operation, further developments are being made to increase the size of fluidised bed combustors (Pruschek *et al.*, 1990), (ERM, CEC, 1995).

### 5.2.2 Description of the Reference Technologies

In FBC systems, fuel is injected and burned in a fluidised bed of particles of mineral mater, usually coal ash, sand or mullite. For coal-firing FBC systems, less than 1 % of the bed material is combustible matter. A part of the combustion air is introduced evenly through the bottom of the bed, which it fluidises at the same time. The remaining air is introduced at a higher level of the furnace, realising an air-staging combustion. The temperature of the bed is normally held within 750-950°C which is the optimum temperature range for $SO_2$ removal and low $NO_x$ emissions, while ensuring good combustion efficiency. The combustion gases from the bed are passed through a conventional heat exchanger in order to reduce the gases to a normal stack exit temperature of 120-150°C (ERM, CEC, 1995).

One of the advantages of FBC systems is that without secondary measures the $SO_2$ and $NO_x$ emissions are lower than for conventional technologies. By injecting limestone into the fluidised bed, sulphur is absorbed and hence $SO_2$ emissions are reduced. Furthermore, due to the low combustion temperature the formation of thermal $NO_x$ is low (ERM, CEC, 1995). However, if the emission standards are further decreased secondary emission reduction measures will become necessary.

FBC can be divided into stationary and circulating FBCs. In stationary FBCs there is either no circulation or only inside the bed of the bed materials. With circulating FBCs the bed materials are circulated externally. The particulates are separated by a cyclone and reintroduced into the bed. Furthermore, FBCs can be distinguished according to whether the combustion takes place under atmospheric pressure or higher. The latter is called pressurised FBC (PFBC). The object of pressurising is to increase the rate at which oxygen is brought into contact with the fuel and to decrease the bed dimensions for a given output (Pruschek *et al.*, 1990), (ERM, CEC, 1995).

The concept of pressurised fluidised bed combustion (PFBC) is suitable for many types of coal. It allows a higher efficiency than the conventional design and the observation of the present emissions standards without secondary mitigation measures. PFBC demonstration plants are currently being operated or installed in the US, Spain, the UK, and Germany. The first commercial plant (70 $MW_{el}$) was commissioned in 1989 at Vartan in Sweden. Since then additional units of about the same size have been commissioned in Spain (ESCATRON, 79 $MW_{el}$) and in the US (Tidd, 135 $MW_{el}$) (ERM, CEC, 1995). Furthermore a unit is under construction at Wakamatsu, Japan. Future units will operate on a larger scale. However, some technical uncertainties have still to be resolved e.g. the reliability of solids feeding and ash removal methods. The estimated cost of a mature PFBC combined cycle design is projected to represent a reduction in capital investment over pulverised coal-firing and savings in the cost of electricity (Blok *et al.*, 1996).

***$N_2O$ emissions from FBC***

Coal, compared to gas, is unique in its ability to produce $N_2O$ emissions. Around 1990, coal combustion was implicated as the main anthropogenic source for $N_2O$ due to errors encountered in sampling (Hayhurst, Lawrence, 1992), (Mann *et al.*, 1992). Corrected measurements have shown that emissions from coal-fired units are typically less than 10–20 mg/$Nm^3$ $N_2O$. Fluidised bed coal combustion is the exception, though, yielding $N_2O$ levels of about 100–250 mg/$Nm^3$ (Kramlich, Linak, 1994), under unfavourable conditions even up to 500 mg/$Nm^3$ (Wójtowicz *et al.*, 1993). This compares with typical levels of $NO_x$ in the range of 200–400 mg/$Nm^3$ from fluidised beds without secondary emission reduction measures (Hayhurst, Lawrence, 1992).

The formation and destruction of $N_2O$ and of NO during coal combustion are closely linked. The complexity of the chemical reactions is still insufficiently understood although great strides towards a better understanding have been made in recent years. Coal-N transformation during combustion can be divided into three stages (Wójtowicz *et al.*, 1993):

- coal devolatilisation and pyrolysis,
- $N_2O$/NO formation (both in the gas phase and at the surface of the char), followed by
- $N_2O$/NO destruction (mainly reduction to $N_2$) by homo- or heterogeneous reactions.

The temperatures in fluidised bed combustors are so low that thermal NO is hardly produced while the thermal decomposition of $N_2O$ is relatively low. In addition, the presence of the solid phase provides a large area for radical recombination, which in turn reduces the rate of removal of $N_2O$ by free radicals. $N_2O$ is formed entirely from fuel-nitrogen, released mainly as HCN and $NH_3$ in purely gaseous flames. The contribution from HCN is more important than that from $NH_3$ (Hayhurst, Lawrence, 1992).

In four major reviews the following conclusions have been drawn concerning the $N_2O$ and $NO_x$ emissions from coal-fired FBC (Hayhurst, Lawrence, 1992), (Mann *et al.*, 1992), (Wójtowicz *et al.*, 1993), (Kramlich, Linak, 1994):

- In FBC the only contribution to final $NO_x$/$N_2O$ emissions comes from fuel-bound nitrogen.

- Temperature is the most important single parameter that controls $N_2O$ levels in an FBC, high temperature leading to reduced $N_2O$ emissions.
- Another important influencing parameter, although less significant, is the fuel type. In general, lower rank fuels tend to yield lower $N_2O$ emissions. This has been attributed to the tendency of the lower rank fuels to favour $NH_3$ release over HCN release. HCN is acknowledged to be more efficiently converted to $N_2O$. Alternatively, the higher surface area of lower rank fuels may promote more complete heterogeneous $N_2O$ destruction.
- An inverse relationship between the oxygen : nitrogen ratio of the fuel and $N_2O$ and an increase with the carbon content of the coal have been observed. Although the specific mechanisms underlying these correlations are unclear, both the oxygen : nitrogen ratio and coal carbon content can, within limits, be indirect indicators of coal rank.
- In experiments where the temperature effect was removed, higher excess air generally increases $N_2O$ emissions. However, excess air has been an unusually difficult parameter to distinguish from temperature because they are so closely coupled in fluidised bed operation.
- The different studies on the influence of limestone addition on $N_2O$ emissions showed a considerable disparity in the results. In all cases however, the change in emission was small, whether the trend was up or down.
- If selective catalytic reduction is employed as secondary $NO_x$ reduction measure in FBC the yield of $N_2O$ varies widely. It depends on catalyst type, catalyst treatment (e.g. crystal size), contamination, support, and background gas composition and likely on catalyst age, too.

In the following the range of $NO_2$ emission in the flue gas of FBC from 100–250 mg/Nm$^3$ is assumed for the analysis. This generous range should cover most of the possible outcomes.

Two PFBC technologies are analysed as reference technologies. The technical specifications of the reference power plants are based on the database IKARUS (Wehovsky *et al.*, 1994). The site of both power plants is assumed to be Lauffen, about 35 km north of Stuttgart.

***The PFBC Plant of 1990***

The first reference technology is a coal-fired combined cycle power plant with PFBC of the present state of technology (1990) (see Figure 5.10 for the system flowsheet of the plant). The generator capacity of the power plant is 181 $MW_{el}$, 33 $MW_{el}$ of the capacity is produced by the gas turbine, 148 $MW_{el}$ by the steam turbine. The electricity sent out is 172.5 $MW_{el}$. The net efficiency of the plant is 42.8 %. Table 5.14 summarises the technical characteristics of the power plant.

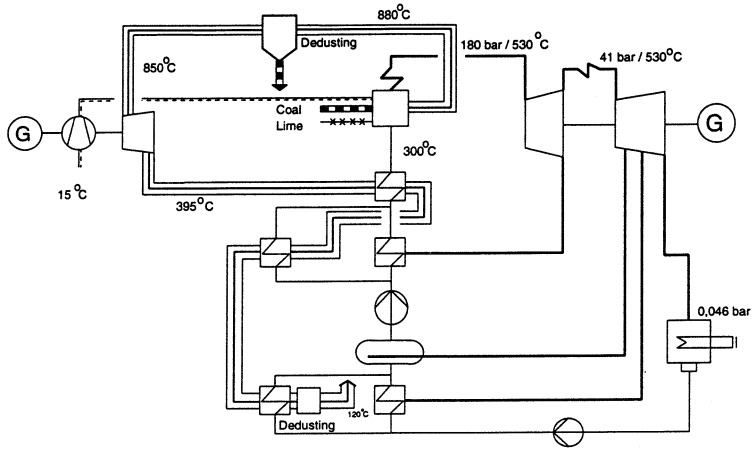



**Figure 5.10** System flowsheet of the coal-fired combined cycle plant with PFBC (state of technology of 1990) (Wehovsky *et al.*, 1994)

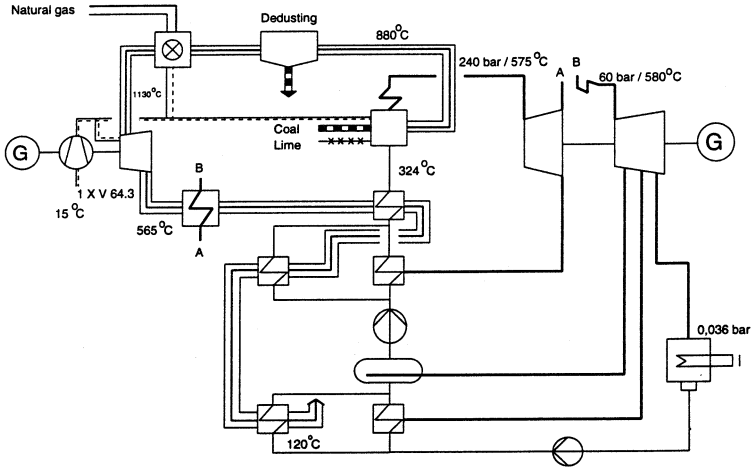



**Figure 5.11** System flowsheet of the coal-fired combined cycle plant with PFBC and additional gas-firing (state of technology of 2005) (Wehovsky *et al.*, 1994)

### *The PFBC Plant of 2005*

The second reference technology is a coal-fired combined cycle power plant with PFBC *and* additional gas firing of the state of technology as predicted for the year 2005 (see Figure 5.11 for the system flowsheet of the plant). The speciality of this combined cycle plant is that with the PFBC water steam (240 bar/575°C) electricity is generated *and* with its flue gases - after additional gas firing - a gas-turbine is driven which increases the process pressure and generates electricity. The generator capacity of the power plant is 196 $MW_{el}$, 60.9 $MW_{el}$ of the capacity is provided by the gas turbine, 135.1 $MW_{el}$ by the steam turbine. The electricity sent out is 187.5 $MW_{el}$. The net efficiency of the plant is 48.5 %. Further technical characteristics are given in Table 5.14.

The inlet temperature of the gas-turbine is increased to 1130°C by the additional gas firing, thus increasing the efficiency and the power share of the gas-turbine. Due to the additional gas firing the $NO_x$ emission factors cannot be safely predicted. Practice will only show if the $NO_x$ emissions will be significantly increased. Here, as a tentative estimate the same flue gas concentrations as for the 1990 technology are assumed for the $NO_x$ and $N_2O$ emission factors. The $CH_4$ emission factor is calculated from the emission factors of coal and gas.

#### *Up- and Downstream Process Steps and Their Emissions*

*The PFBC Plant of 1990*
The PFBC plant of 1990 is coal-fired. The analysis of external effects due to electricity generation from coal takes into account the process steps coal extraction and transport, lime extraction and transport, construction, operation and dismantling of the power plant as well as the disposal of wastes (see Figure 5.12). The upstream processes have been described in detail in the coal fuel cycle report (European Commission, 1995b). All hard coal in Germany is extracted from underground mines. Coal for the Lauffen power plant is obtained from the Ruhrgebiet and the Saarland fields. It is assumed that two thirds of the coal is transported by barge and one third by rail from the coal mines to the power plant. The transport follows the rivers Ruhr, Rhein, and Neckar.

The residues from FBC and their disposal have been analysed extensively (see e.g. (Kautz, 1989), (Gallenkemper *et al.*, 1993), (Nilsson, Clarke, 1994)). FBC produces residues of about the same amount as pulverised coal combustion with FGD. The concentrations of trace elements and of PAH have generally been found to be in the same range as those reported for fly ash from pulverised combustion. The results from various leaching studies indicate that the environmental effects associated with disposal or utilisation of PFBC residues should be no greater than those for fly ash from pulverised combustion of for AFBC residues. However, it will be more difficult as with fly ash or FGD gypsum to produce a residue with a consistent, user-defined specification, which will make it harder to develop utilisation options. Here, it is assumed that the residues are disposed of in a landfill.

**Table 5.14** Technical characteristics of the reference power plants

| | | PFBC 1990 | PFBC 2005 |
|---|---|---|---|
| Generator capacity | [$MW_{el}$] | 181 | 196 |
| Electricity sent out | [$MW_{el}$] | 172.5 | 187.5 |
| Net efficiency | [%] | 42.8 | 48.5 |
| Full load hours per year | | 2602 | 6500 |
| Annual generation | [GWh] | 448.8 | 1218.8 |
| Projected lifetime | [year] | 35 | 35 |
| Floor space | [$m^2$] | 20000 | 21000 |
| *Data relevant for air quality models* | | | |
| Stack height | [m] | 200 | 200 |
| Stack diameter | [m] | 10 | 10 |
| Flue gas volume stream (full load) | [$Nm^3/h$] | 496925.4 | 536038.5 |
| Flue gas temperature | [°C] | 120 | 120 |
| Excess air | | 1.25 | gas turbine: 1.6<br>steam turbine: 1.25 |
| *Fuel specification* | | | |
| Calorific value $H_u$ | [MJ/kg] | 29.2 | 29.2 (coal), 43.6 (gas) |
| Sulfur content | [%] | 0.9 | 0.9 (coal), 0.0 (gas) |
| *Emissions per electricity produced and per flue gas volume stream* | | | |
| $SO_2$ | [mg/kWh] | 580 | 530 |
| | [$mg/Nm^3$] | 200 | 183 |
| $NO_x$ | [mg/kWh] | 430 | 429 |
| | [$mg/Nm^3$] | 150 | 150[1] |
| TSP | [mg/kWh] | 15 | 13 |
| | [$mg/Nm^3$] | 5 | 4.5 |
| $CO_2$ | [g/kWh] | 749 | 608 |
| CO | [mg/kWh] | 430 | 290 |
| $CH_4$ | [mg/kWh] | 42 | 42[1] |
| $N_2O$ | [mg/kWh] | 290–725 | 290–725 |
| | [$mg/Nm^3$] | 100–250[2] | 100–250[1,2] |
| *Bulk materials and wastes* | | | |
| Cooling water | [g/kWh] | 1769 | / |
| Process water | [g/kWh] | 43.6 | / |
| Lime (CaO) | [g/kWh] | 19.7 | 18.1 |
| Ashes | [g/kWh] | 38.3 | 35.2 |
| Waste water | [g/kWh] | 496 | / |

[1] See text for the assumptions used here to allow a preliminary impact analysis.

[2] Flue gas concentrations as listed in the literature (see text); Wehovsky *et al.* assumed 290 mg/kWh based on CORINAR Default Emission Factors (1992), which is regarded as too low.

Source: Wehovsky *et al.* (1994)

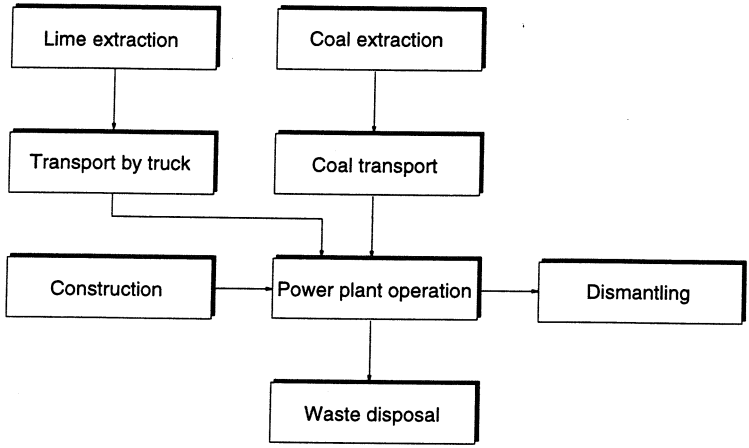



**Figure 5.12** Process steps of the reference energy system of 1990

**Table 5.15** Emissions from the PFBC power plant of 1990 and its upstream processes

| | | Coal mining | Coal preparation | Transports | Power Plant | Total |
|---|---|---|---|---|---|---|
| $SO_2$ | [g/MWh] | 35.3 | n.q. | 3.0 | 580 | 618 |
| $NO_x$ | [g/MWh] | 20.7 | n.q. | 23.9 | 430 | 475 |
| TSP | [g/MWh] | 2.9 | 7.2[2] | 115.2[1]+0.6[3] | 15 | 141 |
| $CO_2$ | [kg/MWh] | 31.3 | n.q. | 3.0 | 749 | 783 |
| CO | [g/MWh] | 2.1 | n.q. | 6.6 | 430 | 439 |
| $CH_4$ | [g/MWh] | 3283.0[2]+0.1[3] | n.q. | 3.5 | 42 | 3329 |
| NMHC | [g/MWh] | 0.6 | n.q. | 2.3 | n.q. | 2.9 |
| $N_2O$ | [g/MWh] | 0.7 | n.q. | 0.002 | 290–725 | 291–726 |

[1] Near surface emissions
[2] Direct emissions
[3] Emissions due to energy demand
n.q. not quantified

The quantification of the emissions from the upstream processes is based on the VDEW 1.0 database for the GEMIS 2.1 model (Fritsche *et al.*, 1994), (FDE, 1996). Table 5.15 summarises the emissions from the PFBC plant of 1990 and its upstream processes.

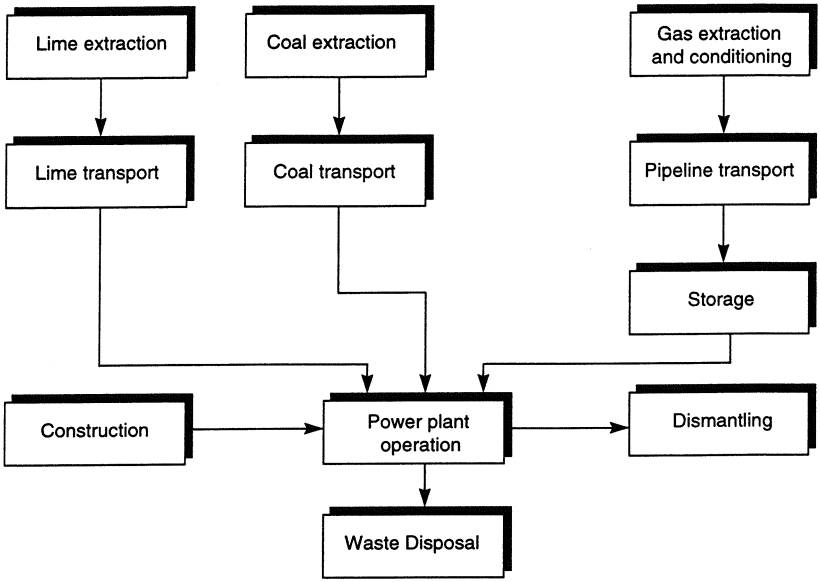



**Figure 5.13** Process steps of the reference energy system of 2005

*The PFBC Plant of 2005*
The PFBC plant of 2005 uses two fuels: coal and natural gas (see Figure 5.13). The emissions of the coal fuel cycle have been assessed as for the 1990 PFBC power plant. The gas fuel cycle is described for the gas-fired condensing boiler in section 3.1. The emissions are quantified accordingly. Again, direct emissions from gas extraction, conditioning and transport in the GUS have been quantified with the high leakage rates given in VDEW 1.0 as well as with the lower leakage given in (Frischknecht *et al.*, 1996). Table 5.16 summarises the emissions from the PFBC plant of 2005 and its upstream processes. It is noteworthy, that despite its higher efficiency, the $SO_2$ and $NO_x$ emissions of the upstream processes of this PFBC plant are higher than that for the less modern plant because of the high direct emissions during gas extraction and conditioning.

As for the conventional fossil power plants most of the air-borne emissions of the two PFBC reference energy systems are due to the power plant itself - with few exceptions. The exceptions are

- the direct particulate emissions due to loading and unloading during coal transport and preparation; these emissions have only a local impact and are not taken into account in the impact assessment;
- the high direct $CH_4$ and NMHC emissions during coal mining, gas extraction, conditioning and transport.

**Table 5.16** Emissions from the PFBC power plant of 2005 and its upstream processes (maximum leakage in gas extraction and conditioning in the GUS assumed)

| | | Extraction | Preparation / Conditioning | Transports | Power Plant | Total |
|---|---|---|---|---|---|---|
| $SO_2$ | [g/MWh] | | | | 530 | 593 |
| • Coal | | 22.3 | n.q. | 1.9 | | |
| • Gas | | 6.9[2]+0.1[3] | 30.6[2]+1.0[3] | 0.02 | | |
| $NO_x$ | [g/MWh] | | | | 429 | 506 |
| • Coal | | 13.2 | n.q. | 15.2 | | |
| • Gas | | 4.0[2]+6.6[3] | 0.1[2]+3.5[3] | 13.1 | | |
| TSP | [g/MWh] | | | | 13 | 97 |
| • Coal | | 1.9 | 4.6[1] | 73.1[1]+0.4 | | |
| • Gas | | 0.1 | 5.7 | 0.2 | | |
| $CO_2$ | [kg/MWh] | | | | 608 | 635 |
| • Coal | | 19.8 | n.q. | 1.9 | | |
| • Gas | | 0.7[2]+1.4[3] | 1.3 | 2.2 | | |
| CO | [g/MWh] | | | | 290 | 307 |
| • Coal | | 1.3 | n.q. | 4.2 | | |
| • Gas | | 3.9 | 1.8 | 6.1 | | |
| $CH_4$ | [g/MWh] | | | | 42 | 2673[4] |
| • Coal | | 2084.0[2]+0.07[3] | n.q. | 2.2 | | |
| • Gas | | 281.5[2]+0.5[3] | 150.3[2]+1.8[3] | 112.2[2]+0.3[2] | | |
| NMHC | [g/MWh] | | | | / | 23.5[5] |
| • Coal | | 0.4 | n.q. | 1.5 | | |
| • Gas | | 20.0[2]+0.3 | 0.2[2]+0.2[3] | 0.2[2]+0.7[3] | | |
| $N_2O$ | [g/MWh] | | | | 290–725 | 291–726 |
| • Coal | | 0.4 | n.q. | 0.001 | | |
| • Gas | | 0.1 | | 0.1 | | |

1 Near surface emissions

2 Direct emissions

3 Emissions due to energy demand

4 Total of direct emissions from the gas fuel cycle assuming maximum leakage is 544 g $CH_4$ per MWh; assuming minimum leakage total of direct emissions is 225 g/MWh, the total of direct and indirect emissions from the coal and gas fuel cycle 2353 kg/TJ, then.

5 Total of direct emissions from the gas fuel cycle assuming maximum leakage is 20 g NMHC per MWh; assuming minimum leakage total of direct emissions is 4 g/MWh, the total of direct and indirect emissions from the coal and gas fuel cycle is 7.5 g/MWh, then.

n.q. not quantified

### 5.2.3 The Priority Impact Pathways

For the two PFBC reference energy systems the same impact pathways are important as for the other fossil energy systems (European Commission, 1995b), (European Commission, 1995c):

- Effects of atmospheric pollution on
  - human health,
  - crops,
  - forests and unmanaged ecosystems,
  - materials
- occupational health effects, and
- impacts of global warming.

Ozone layer depletion is often referred to as a relevant impact of coal-fired FBC (see e.g. (Hayhurst, Lawrence, 1992), (Wójtowicz *et al.*, 1993), (Kramlich, Linak, 1994)) because of its comparatively high $N_2O$ emission factor (see p. 137). $N_2O$ influences the depletion of the ozone layer indirectly via the decomposition of $N_2O$ in the stratosphere, which is an important source of NO and $NO_2$ molecules there (Fabian, 1992). On the one hand, stratospheric NO is one of the important catalysts of the decomposition reactions of stratospheric $O_3$, on the other hand, $NO_2$ serves as a temporary sink for the ozone depleting chlorine monoxide (ClO); the net effect is uncertain (Worrest *et al.*, 1989), (Kramlich, Linak, 1994). In the second assessment report of the IPCC stationary combustion of fossil fuels is no longer listed as one of the important anthropogenic sources of $N_2O$ emissions (IPCC, 1996). Thus, ozone layer depletion is not regarded as *priority* impact pathway, here.

### 5.2.4 The Quantification of Damage Costs

The impacts and damages of the two fuel cycles under analysis were quantified following the EXTERNE methodology established by now. Appendix I to VII summarise the approaches, exposure-response functions and monetary values used and describe the software tool EcoSense employed for the assessment.

As site-specific ozone models are not available at the moment, ozone impacts are assessed using the damage costs derived by Rabl and Eyre (1997) with a simplified approach. In this approach results for the ozone models of EMEP and Harwell were combined with the exposure-response functions and economic valuation for health and agricultural crops recommended in (European Commission, 1995a) to derive average damage factors for $NO_x$, NMHC and $CH_4$ emissions in Europe (incl. GUS).

In Table 5.17 the quantified damages for the two fuel cycles are summarised. The damage costs due to air emissions of the upstream process steps have been assessed by assuming that the emissions occur at the power plant site. Employing the YOLL approach for mortality effects, for the PFBC reference system of 1990 damage costs (without global warming) of 16.6 mECU/kWh were quantified, employing the VSL approach the damages increase to 60.9 mECU/kWh. The respective results for the PFBC plant of 2005 are 15.9 mECU/kWh and

59.1 mECU/kWh. Increased mortality, mostly due to sulphate and nitrate aerosols, contributes the most to these results. Due to the high direct $SO_2$ and $NO_x$ emissions during gas extraction and conditioning the air pollution effects of the upstream processes of the PFBC2000 system are higher than those of the 1990 system.

Employing the EXTERNE results (illustrative restricted range) for global warming (see Appendix V) the global warming damages of the two reference energy systems are about the same order of magnitude as all the other quantified damages together. For the PFBC1990 systems global warming damages of about 17–43 mECU/kWh are quantified assuming the lower $N_2O$ emission factor. These values increase to 19–49 mECU/kWh for the higher $N_2O$ emission factor. This shows, that although the uncertainty about the $N_2O$ emissions of FBC has a noticable influence on the results, it is still small compared to the other uncertainties attached to the quantification of the global warming costs. The global warming damage costs for the PFBC2005 system assuming the lower $N_2O$ emissions are 14–36 mECU/kWh.

Table 5.18 summarises the totals of the quantified damages (based on the lower $N_2O$ emission factor), depending on the choice of YOLL/VSL for mortality effects and on the global warming damage costs per tonne $CO_2$. Taking into account these sensitivities the quantified total costs range from 20 to 192 mECU/kWh for the PFBC 1990 system and from 19 to 168 mECU/kWh for the PFBC 2005 system.

### 5.2.5 Conclusions

Fluidised bed combustion is a comparatively new thermodynamic cycle for electricity production. One of its advantages is that without secondary measures the $SO_2$ and $NO_x$ emissions are lower than for conventional technologies. However, if the emission standards are further decreased secondary emission reduction measures will become necessary. Furthermore, the $N_2O$ emissions of coal-fired FBC are one order of magnitude higher than those of pulverised coal combustion. This fact has created some concern because $N_2O$ contributes to global warming and influence ozone-depletion. Nonetheless, ozone depletion is not identified as *priority* impact pathway of FBC energy systems because at the moment the net effect is unclear and FBC still is not an important anthropogenic source of $N_2O$.

Two circulating PFBC technologies were selected as reference technologies. The first is a coal-fired combined cycle power plant with PFBC of the present state of technology. Its net capacity is 181 MW. The second is a coal-fired combined cycle power plant with PFBC and additional gas firing as predicted for the year 2005 (net capacity 188 MW). The site of the two reference power plants is assumed to be Lauffen, 35 km north of Stuttgart.

The emissions of the upstream processes of the reference energy systems were quantified based on the GEMIS 2.1 model employing the VDEW 1.0 database. Thereby, despite the higher net efficiency of the more modern second technology, the $SO_2$ and $NO_x$ emissions of the upstream processes are higher than those of the upstream processes of the plant of 1990 because of the high direct emission during gas extraction and conditioning.

**Table 5.17** Damages of the PFBC power plants and their upstream processes (negative numbers constitute benefits)

| | PFBC 1990 | PFBC 2005 | $\sigma_g$ |
|---|---|---|---|
| *POWER GENERATION* | | | |
| Public health | | | |
| Mortality – YOLL[1] *(VSL)* | 11.6 *(48.3)* | 10.8 *(44.3)* | B |
| *of which* *TSP* | 0.2 *(0.7)* | 0.2 *(0.7)* | |
| *$SO_2$* | 6.3 *(28.7)* | 5.7 *(25.5)* | |
| *$NO_x$* | 5.1 *(18.8)* | 4.9 *(18.1)* | |
| Morbidity | 1.5 | 1.3 | A |
| Accidents | n.q. | n.q. | A |
| Occupational health (see below) | | | A |
| Ozone impacts (human health & crops) – YOLL[1] *(VSL)* | 0.7 *(4.9)* | 0.7 *(4.8)* | |
| Crops | -0.02 | -0.03 | B |
| Ecosystems | iq | iq | B |
| Materials | 0.2 | 0.2 | B |
| *Monuments* | | | |
| Noise | ng | ng | |
| Visual impacts | ng | ng | |
| Global warming[2] | | | C |
| $CO_2$ and $CH_4$ | | | |
| Conservative 95 % confidence interval | 2.8–104.2 | 2.3–84.6 | |
| Illustrative restricted range | 13.5–34.5 | 11.0–28.0 | |
| $N_2O$ – low emission factor (high in brackets) | | | |
| Conservative 95 % confidence interval | 0.3–12.5 (0.8–31.0) | 0.3–12.5 (0.8–31.0) | |
| Illustrative restricted range | 1.6–4.1 (4.0–10.3) | 1.6–4.1 (4.0–10.3) | |
| *OTHER FUEL CYCLE STAGES* | | | |
| Public health – YOLL[1] *(VSL)* | 1.1 (4.5) | 1.8 (6.7) | |
| Occupational health | 0.9 | 0.6 | |
| Ozone impacts (human health & crops) – YOLL[1] *(VSL)* | 0.5 *(0.5)* | 0.4-0.5 *(1.0-1.1)* | |
| Crops | -0.002 | 0.04 | B |
| Ecological effects | ng | ng | B |
| Materials | 0.02 | 0.03 | B |
| Road damages | ng | ng | A |
| Global warming[2] | | | C |
| Conservative 95 % confidence interval | 0.4–14.3 | 0.3–11.4 | |
| Illustrative restricted range | 1.9–4.7 | 1.5–3.8 | |

[1] YOLL = mortality impacts based on 'years of life lost' approach; here results are calculated assuming a discount rate of 3 %, VSL= impacts evaluated based on 'value of statistical life' approach.

[2] $CO_2$ equivalent emissions are calculated from the $CH_4$ and $N_2O$ emissions employing the Global Warming Potentials of IPCC (1996a) for the time horizon of 100 years.

ng: negligible; nq: not quantified; iq: only impact quantified; - : not relevant

**Table 5.18** Sub-total damages of the PFBC power plants and their upstream processes (low $N_2O$ emission factor is assumed for generation stage)

| YOLL *(VSL)* | PFBC 1990 | PFBC 2005 |
|---|---|---|
| Conservative 95 % confidence interval | 20.1–147.7 *(64.5–192.0)* | 18.8–124.5 *(62.1–167.7)* |
| Illustrative restricted range | 33.5–59.9 *(77.9–104.3)* | 29.9–51.8 *(73.2–95.6)* |

The impacts and damages of the two reference energy systems were quantified following the EXTERNE methodology established by now. As with all the other fossil fuel cycles, increased mortality, mostly due to sulphate and nitrate aerosols, contributes the most to the total quantified damage costs (without global warming). The present results for global warming are about the same order of magnitude as all the other damage costs quantified together, the uncertainty of the global warming damages, however, is especially high. The uncertainty about the exact amount of the $N_2O$ emissions adds to the total uncertainty concerning the global warming damage costs but does not change the overall picture. Using the so-called 'illustrative restricted range' EXTERNE results for global warming, the lower $N_2O$ emission factors and the YOLL-approach for the valuation of mortality impacts, the following costs were quantified:

- PFBC 1990 system: 34–60 mECU/kWh;
- PFBC 2005 system: 30–52 mECU/kWh.

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## 5.3 Fuel Cells for Stationary Applications

### 5.3.1 Introduction

Unlike conventional thermal engines, fuel cells generate direct current electricity and heat by combining fuel and oxygen in an electrochemical reaction. Hence, fuel cell efficiencies are not limited by the Carnot factor as with the (conventional) Rankine cycle technologies because there is no need for an intermediate combustion step and boiling of water. Thus, theoretical fuel-to-electricity conversion efficiencies lie in the range of 80–83% for low temperature fuel stacks and fall to between 78–73% for high temperature stacks. In practice, low temperature fuel cells with hydrogen as fuel achieve fuel-to-electricity system efficiencies of 50–60% while for high temperature fuel cells fuelled by natural gas efficiencies range from 42% up to 60–65%. Thus, fuel cells have the potential to contribute in a significant way to emission reduction, especially of $CO_2$ and $NO_x$ (Blok *et al.*, 1995).

Most fuel cells are still in the development stage. Potential markets are the power sector, combined heat and power (CHP) systems and transportation. However, it is doubtful that fuel cells will ever become attractive for power generation only because of the high capital costs. Medium and high temperature cells may become attractive for industrial CHP while low temperature cells fuelled by hydrogen may be applied in transportation (Blok *et al.*, 1995). The objective of this study was the analysis of stationary applications of fuel cells.

In the following the state of development of fuel cells is further described and the reference energy system defined (section 5.3.2). The air-borne emissions of the reference energy systems are quantified (section 5.3.3) and the impacts and damage costs assessed (section 5.3.5). The section on fuel cells is concluded with a short excerpt on the hydrogen economy (section 5.3.6).

### 5.3.2 The reference technology

***Principle of operation***

A fuel cell consists of a fuel electrode (anode) and an oxidative electrode (cathode), which are separated by an ion-conducting electrolyte. The electrodes are connected by a metallic external circuit to a load. In the metallic part of the circuit, electricity is transported by a flow of electrons, whereas it is transported inside the electrolyte by a flow of ions, like the hydrogen ion ($H^+$) or the hydroxyl ion ($OH^-$) in acid or alkaline electrolytes respectively. In the case of molten carbonate electrolytes, the corresponding ionic carriers are carbonate ions ($CO_3^{2-}$), in solid oxide systems the carriers are oxide ions ($O^{2-}$).

Theoretically, any substance capable of chemical oxidation and being able to be supplied continuously (like a fluid) can be burned galvanically at the anode of a fuel cell. Just as, the oxidant can be any fluid, that can be reduced at a sufficient rate. Due to its high reactivity, gaseous hydrogen or hydrogenous gas is used in fuel cells in most cases. Similarly the most com-

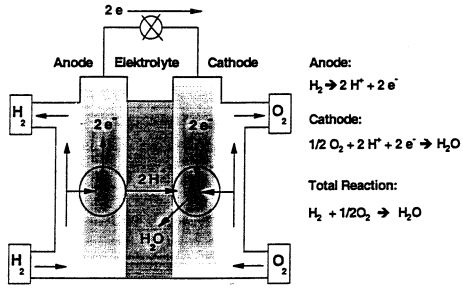



**Figure 5.1** Principle of operation of a fuel cell

mon oxidant at the fuel cell cathode is gaseous oxygen which is readily and economically available from the air.

In spite of the complexities involved in the construction and operation of an applicable fuel cell system, the principle of operation of a fuel cell is already understood. Figure 5.1 shows a hydrogen-oxygen fuel cell employing an acid electrolyte in a simplified and schematic way. At the anode, incoming hydrogen gas ionises to produce hydrogen ions and electrons. Since the electrolyte is a non-electronic conductor, the electrons run off the anode via the metallic external circuit. At the cathode, oxygen gas reacts with the migrating hydrogen ions of the electrolyte and electrons emerging from the external circuit. Depending on the operating temperature of the cell, the produced water enters the electrolyte, diluting it and increasing its volume, or it leaves the cell trough the cathode as vapour. In all fuel cells with liquid electrolytes operating below the boiling point of water, a careful water management is necessary to remove the produced water (Appleby, 1989).

*Types*

The different types of fuel cells are characterised by the level of the working temperature and the composition of the electrolyte. Table 5.19 enables an overall view of the most important types. In the field of the low temperature fuel cells, which have a working temperature of about 80°C, two important types of fuel cells exist today: the Alkaline Fuel Cell (AFC) and the Solid Polymer Fuel Cell (SPFC). We assume here for our reference cases that both types operate with pure oxygen as the oxidant gas. Medium-temperature fuel cells work in the range 150°C to 300°C using air as the oxidant gas. Complete CHP plants fitted with this type of fuel cell are already commercially available. The working temperature of high-temperature fuel cells are in the range of 600°C to 1000°C. In this area two important types of fuel cells exist today: the Molten Carbonate Fuel Cell (MCFC) and the Solid Oxide Fuel Cell (SOFC).

The realisation of an adequate current density and tolerable voltage losses inside the fuel cells needs special electrode materials and catalysts which restrict the selection of a suitable fuel. Gaseous hydrogen has become the major used fuel. Hydrogen can be generated of natural gas which consists of 90% of methane ($CH_4$). The hydrogen can be provided by a steam-

reforming process outside the fuel cell ($CH_4 + H_2O \rightarrow 3\ H_2 + CO$) with a following carbon-monoxide shift ($CO + H_2O \rightarrow H_2 + CO_2$). High-temperature fuel cells are also capable of internal methane reforming, which reduces the expenditure for the preparation of the burning gas and increases the efficiency of the whole system.

Usually gasified oxygen or air is used as oxidant. The selection depends on whether the electrolyte and the electrodes are compatible with the share of inert gas in the air. We assume for the cases considered here that low-temperature fuel cells are operated by pure oxygen and that the medium- and high-temperature fuel cells are run using air. Further more a low carbon monoxide concentration in the burning gas is necessary for the low and medium-temperature fuel cells.

There are better chances for a commercial application in the energy sector for medium and high temperature fuel cells. In the following sections the technical concepts of PAFC and MCFC fuel cell plants are described. Owing to the low stage of development of the SOFC, this technology will not be considered further.

**Table 5.19.** Technical data of fuel cell systems (Bine, 1992; Daun, 1992; Knappstein, 1993a; Knappstein 1994; Roesler, 1992; Eichenberger, 1994).

| | Unit | Low-Temperature Fuel Cell | | Medium-Temperature Fuel Cell |
|---|---|---|---|---|
| | | **Alkaline Fuel Cell AFC** | **Solid Polymer Fuel Cell SPFC** | **Phosphoric Acid Fuel Cell PAFC** |
| | | ***General Characteristics*** | | |
| Development stage and perspectives | | This type of fuel cell was often used in space operations. A condition for an expanded commercial application is a more cost-effective manufacturing process and a more economical use of precious metals. | This type of fuel cell was not often used in the past. A potential of technical development exists. A condition for an expanded commercial application is a more cost-effective manufacturing process, a more economical use of precious metals and cheaper electrolytes. | At present there is no development of this type of fuel cell in Europe. In the USA and Japan commercial prototypes of fuel cell plants are already available. They operate world-wide at test plants. 1991 a 11-$MW_{el}$ power plant was put into operation in Japan. In Europe at present about 3,3 $MW_{el}$ are installed mainly in smaller cogeneration plants with an electric power of about 50 $kW_{el}$ to 200 $kW_{el}$ at present. |
| Application | | space operation, submarine, traffic | space operation, submarine, traffic | cogeneration, power generation |

| | Unit | Low-Temperature Fuel Cell | | Medium-Temperature Fuel Cell |
|---|---|---|---|---|
| | | Alkaline Fuel Cell (AFC) | Solid Polymer Fuel Cell (SPFC) | Phosphoric Acid Fuel Cell (PAFC) |
| Availability | | | | |
| • pattern | | 1991 | 1991 | 1991 |
| • for power station service | | - | - | 1992–1994 |
| *Technological specification* | | | | |
| Fuel | | gaseous hydrogen | gaseous hydrogen | natural gas, methane, gaseous hydrogen |
| Oxidant gas | | pure oxygen | pure oxygen | air |
| Electrolyte | | KOH | sulphur acid, fixed in per-fluorinated polymer | $H_3PO_4$, fixed in a matrix |
| Ionic conduction in the electrolyte | | hydroxyl ions | hydrogen ions | hydrogen ions |
| Electrode materials | | Ag/Ni or Pt/Pt on PTFE | Pt or Pt-$RuO_2$ | Pt |
| Working temperature | °C | 80 | 80 | 190 |
| Cell voltage | V | 0.75 | 0.72 | 0.6 – 0.7 |
| Current density | $A/cm^2$ | 420 | 500 | 250 |
| Cell-efficiency (full load) | | | | |
| • electric | % | 63 | 50–60 | 45 |
| • thermal | % | - | - | 30 |
| Service life (cell) | h | 2 000 | 20 000 | > 10 000 |
| Unit size | | | | |
| • prototype | $kW_{el}$ | < 20 | 20 | 40, 200, 4 500 |
| • commercial type | $kW_{el}$ | - | - | 200, 1 MW, 5 MW, 11 MW |
| Aimed system size | $kW_{el}$ | 100 | 100 | 1–11 MW |
| *Costs* | | | | |
| Spec. investments | | | | |
| • at present | ECU/$kW_{el}$ | 27 500 | 27 500 | 2 450 |
| • planned | ECU/$kW_{el}$ | 2 300 | 2550 | 1 300 |
| *Manufacturer* | | | | |
| • Prototype | | Siemens, Elenco, IFC ERC | Siemens, Ballard, ABB, Hamilton | ICF (ONSI), FUJI, Toshiba, MBB, Ansaldo, Westinghouse |
| • Commercial type | | Siemens, Elenco, IFC, ERC | Hamilton | ICF (ONSI), FUJI, Toshiba |

**Table 5.19** continued

| | Unit | High-Temperature Fuel Cell | |
|---|---|---|---|
| | | Molten Carbonate Fuel Cell MCFC | Solid Oxide Fuel Cell SOFC |
| | | *General Characteristics* | |
| Development stage and perspectives | | Intensive research and development activities in the USA, Japan, Holland and Germany. 100 $kW_{el}$ are the largest realised units (ERC, USA) at the moment. In Santa Clara (USA) a 2 $MW_{el}$ Power plant is built. | Research and development activities in the USA and Germany. This type of fuel cell has got a low development stage, about 40 $kW_{el}$ are realised in laboratory stage at present. |
| Application | | cogeneration, power generation | cogeneration, power generation |
| Availability | | | |
| • pattern | | 1991 | 1995 |
| • for power station service | | 2000–2020 | 2005–2020 |
| | | *Technological specification* | |
| Fuel | | natural gas, methane, gaseous hydrogen | natural gas, methane, gaseous hydrogen |
| Oxidant gas | | air | air |
| Electrolyte | | $Li_2CO_3/K_2CO_3$ or.$Li_2CO_3/Na_2CO_3$ | $ZiO_2$-diaphragm, Y-dotiert |
| Ionic conduction in the electrolyte | | CO-ions | oxygen ions |
| Electrode materials | | $NiOLi_x$ or $LiFeO_2$ | $LaNiO_3/LaMnO_3$ Ni-Co-ceramics |
| Working temperature | °C | 650 | 1000 |
| Cell voltage | V | 0.7–0.9 | 0.6 |
| Current density | $A/cm^2$ | 100–200 | 250–400 |
| Cell-efficiency (full load) | % | 50–55 | >50 |
| • electric | % | 42 | 40 |
| • thermal | | | |
| Service life (cell) | h | 10 000 | 16 000 |
| Unit size | | | |
| • prototype | $kW_{el}$ | 10 | 3–25 |
| • commercial type | $kW_{el}$ | - | - |
| Aimed system size | $kW_{el}$ | 5–50 MW | 10–200 |
| | | *Costs* | |
| Spec. investments | | | |
| • at present | ECU/k $W_{el}$ | 10 000 | 5 250 |
| • planned | ECU/k $W_{el}$ | 850 | 1 000 |
| | | *Manufacturer* | |
| • Prototype | | IHI, FUJI, Toshiba, ERC, Ansaldo, Hitachi | Westinghouse, ETL, Dornier, Argonne |
| • Commercial type | | Hitachi, MBB, Mitsubishi | |

***PAFC CHP Station(Reference System)***

The PAFC is the fuel cell with the highest stage of development. In the USA and Japan plants with a total generator capacity of several $MW_{el}$ are in operation. In Europe about 15, mainly smaller cogeneration plants are installed for demonstration and testing the PAFC technology. At present, only the US-manufacturer ONSI offers 200 $kW_{el}$ commercial fuel cell plants, which are designed for operating in cogeneration mode.

Commercial PAFC CHP plants reach an electric cell-efficiency of about 55 %. Losses at the natural gas reformer and the fuel-cleaning unit lead to an electric efficiency of the whole system of about 35 % to 40 %. Due to these high reductions of the efficiency caused by the fuel converting technologies of the PAFC CHP station, it cannot compete with large combined cycle power plants. Therefore, the range of application of PAFC systems for stationary energy conversion will in all probability be in the field of small unit cogeneration (power range 0.1 to 5 $MW_{el}$) (Roesler, 1992). The total efficiency of modern PAFC plants, which means the efficiency of producing electricity and heat, ranges between 80 % to 85 % (Altfeld, 1990). Table 5.20 shows the PAFC plants which are installed in Europe at present.

Figure 5.2 shows the main components and the functional principle of the 200 $kW_{el}$ fuel cell total energy unit PC25 from ONSI. In this plant the natural gas is converted into a hydrogenous process gas. For that, the natural gas must be purified of sulphur in a first step. Than it has to be mixed with steam of 800 °C and fed into the reforming process. In a further step the carbon monoxide will be converted into carbon dioxide and hydrogen at a temperature of approx. 400 °C. The resulting process gas contains about 77 % hydrogen, 20 % carbon dioxide and traces of methane, nitrogen and carbon monoxide. The concentration of carbon monoxide in the fuel gas is limited to 1 % due to the carbon monoxide sensitive platinum catalyst. In the anode of the fuel cell stack, the process gas will be converted to exhaust steam by oxygen. This steam is then fed back to the reforming process. Usually no further feeding of water is necessary to satisfy the steam consumption of the reforming process.

In the case of the PC25, the fuel cell stack generates a constant voltage of 200 V and a continuous current of 1050 A which is transformed into an alternating current. The quality of the current and the frequency must be prepared according to the standards of the mains supply.

The 200 $kW_{el}$ power plant PC25 from the manufacturer ONSI was selected as reference system. Table 5.21 shows the technical data of this fuel cell CHP plant. It operates in the cogeneration mode with a total efficiency of 85 %. Electricity will be generated with an efficiency of 40 %. For the fuel cell stack, a lifetime of about 40 000 h is aimed while a lifetime of about 20 000 h is proved today (Knappstein, 1996). The reference plant is assumed to be located in Lauffen, 35 km north of Stuttgart.

Although there is no detailed list of materials used in the PC25 available from the manufacturer, there can be assumed, that - except for the stack - the fuel cell CHP station consists of conventional power plant technology, mainly steel and high-grade steel. Lower

of remaining data. The total weight of the PC25 is 18 t, that of the cooling unit 0,7 t. With a estimated weight of the stack of 3 t it can be assumed, that for reformer, converter, inverter and the remaining periphery of the PC25 (without cooling unit) about 15 t of steel (7 tons of high grade steel and 8 tons of steel) are used.

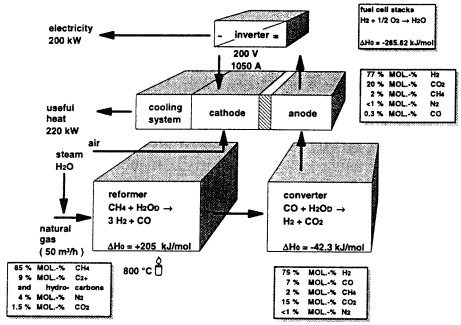



**Figure 5.2** Main components of fuel cell district heating and CHP station PC25

**Table 5.20** PAFC plants in Europe (Stahl, undated)

| Operator | Nation | Manufacturer | Power [$kW_{el}$] | Location | Starting |
|---|---|---|---|---|---|
| Sydkraft | Sweden | FUJI | 50 | Ästrop | 12/1991 |
| Sydkraft | Sweden | ONSI | 200 | Bara | 06/1992 |
| Vattenfall | Sweden | FUJI | 50 | Varberg | 11/1992 |
| Ruhrgas | Germany | ONSI | 200 | Dorsten | 09/1992 |
| HEAG | Germany | ONSI | 200 | Darmstadt | 06/1993 |
| HGW/HEW | Germany | ONSI | 200 | - | - |
| Thyssengas | Germany | ONSI | 200 | Duisburg | 10/1992 |
| Naturgas Syd Sonderjyllands | Denmark | ONSI | 200 | Toftlund | 11/1992 |
| Imatran Voima | Finland | ONSI | 200 | Tavastehus | 12/1992 |
| Austrian Ferngas | Austria | ONSI | 200 | Wien | 01/1993 |
| Enagas | Spain | FUJI | 50 | Madrid | 12/1991 |
| SNAM Enirichere | Italy | FUJI | 50 | Mailand | 11/1991 |
| Aem | Italy | Ansaldo/IFC | 1 300 | Mailand | - |
| Ansaldo/CLC | Italy | ONSI | 200 | Bologna | 04/1993 |
| SIG | Swiss | ONSI | 200 | Genf | 03/1993 |

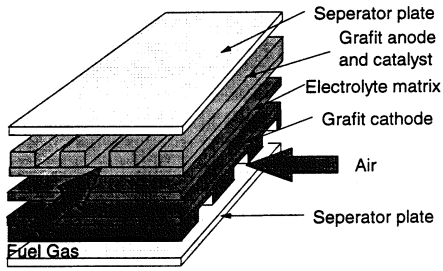



**Figure 5.3** Configuration of phosphoric acid fuel cells

Figure 5.3 shows the principle of construction of a single cell, which is about 5 mm thick. Carbon is the vital material for the electrochemical cell stack: high surface area powder for the support of the electrocatalyst, porous carbon paper for the electrode substrate and graphite for the separator plate. The porous electrode consists of an electrocatalyst layer, i.e. carbon-supported platinum electrocatalysts bonded with teflon and a carbon paper substrate. A braiding matrix between the electrodes consisting of teflon-bonded silicon carbide contains the gel-like phosphoric acid. The matrix is micro-porous and thinly spread on the cathode (Blomen, 1993). The stack of the PC 25 consists of 319 single cells which a total surface of 200 $m^2$. It is loaded with about 200 kg phosphoric acid (Brammer, 1996), the weight of carbon is estimated to be about 20 kg. Furthermore, about 1.5 kg platinum is used in the stack (Kah, 1996). Most of the stack consists of high grade steel (2 800 kg), which is almost entirely located in the periphery (e.g. cooling coils, case and frame). To a lesser degree other materials like teflon or plastics are used in the stack, which cannot be taken into due to lack of data (Appleby, 1989).

***MCFC CHP Station***

In contrast to the PAFC, the MCFC is on a distinctly lower development stage. Most advanced in this technology are the USA and Japan. Headed by MTU Friedrichshafen (Germany) a European syndicate works on the commercialisation of the MCFC. Besides MTU the syndicate consists of two other German partners, Ruhrgas AG and RWE AG, and two Danish partners, Haldor Topsoe A/S and Elkraft A. m. b. A. The activities of this syndicate are based on a licence agreement with the US-manufacturer Energy Research Corporation (ERC).

Figure 5.4 shows the complete development program of the European syndicate. At the end of 1997 the first 200 $kW_{el}$ prototypes are assumed to operate in field tests. The commercialisation of the MCFC is planned for the year 2001 (Kraus, 1995). In the USA a 2 $MW_{el}$ power plant is under construction in Santa Clara at the moment. No detailed information is available about the Japanese state of the art.

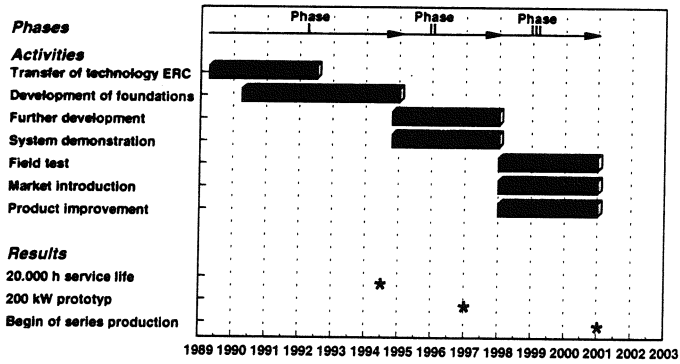



**Figure 5.4** Development program of the European MCFC research

The electrolyte of the MCFC is made of molten carbonate. It is possible to use either carbon monoxide or hydrogen as fuel. Both gases are typical components of the reformer process, therefore a carbon monoxide converter is not necessary. Owing to the high working temperature of about 600°C to 700°C it is possible to reform the fuel gas inside the fuel cells (internal reformation). If the waste heat, which is at a temperature level of about 500°C is not used, the MCFC plant achieve an electrical efficiency between 50% and 60%. Electrical efficiencies up to 65% are attainable, if the waste heat is used for power generation as well. Due to the high temperature level of the waste gas, it can be used to provide process or space heating. In this case, total efficiencies up to 85% can be achieved (Stimming, 1993), (Wendt, 1990).

Similar to the PAFC-CHP plants, detailed data about the materials used to built the CHP plants are not available. According to (Huppmann, 1996) roughly estimated the MCFC-CHP plant has a total weight of 8 tons, 3 tons steel, 4 tons high-grade steel and 1 ton nickel, which is used as electrode material and as catalyst. Other materials like molten carbonate and aluminium are not taken into account here because the respective data is not available.

Table 5.21 summarises the technical data of the MCFC-CHP plant. Analogous to the PAFC-CHP plant it operates in the cogeneration mode with a total efficiency of 77%. Electricity will be generated with an efficiency of 55%. The lifetime of the fuel cell stack is projected to be about 40 000 h. The full load hours are assumed to be 5 000 h per year. Like the PAFC-CHP plant, the MCFC-CHP plant is assumed to be located in Lauffen.

**Table 5.21** Technical characteristics of the reference CHP plants

| Plant type | | PAFC (PC25 from ONSI) | MCFC |
|---|---|---|---|
| Electricity send out | | 200 $kW_{el}$ / 235 kVA | 248 $kW_{el}$ |
| Net efficiency | | 40% | 54% |
| Total efficiency | | 85% | 77% |
| Full load hours per year | | 5000 h | 5000 h |
| Annual generation | | 1000 $MWh_{el}$ | 1240 $MWh_{el}$ |
| Projected lifetime | | 40 000 h | 40 000 h |
| Floor space | | 22 $m^2$ | 22 $m^2$ |
| *Data relevant for air quality modelling* | | | |
| Stack height | | 10 m | 10 m |
| Stack diameter | | 0.5 m | 0.5 m |
| Flue gas volume stream | | 1200 $Nm^3/h$ | 1200 $Nm^3/h$ |
| Flue gas temperature | | 100°C | 100°C |
| *Fuel specification* | | | |
| Calorific value $H_u$ | | 43.6 MJ/kg | 43.6 MJ/kg |
| Sulfur content | | 0% | 0% |
| *Emissions per electricity produced and per flue gas volume stream* | | | |
| $SO_2$ | [kg/a] | 0 | 3 |
| | [mg/$Nm^3$] | | 0.4 |
| $NO_x$ | [kg/a] | 3 | 20 |
| | [mg/$Nm^3$] | | 3.5 |
| TSP | [mg/$Nm^3$] | 0 | 0 |
| $CO_2$ | [t/a] | 496 | 460 |
| CO | [kg/a] | 6 | 6.6 |
| | [mg/$Nm^3$] | 37 | 8 |
| $CH_4$ | [kg/a] | 37 | 0 |
| | [mg/$Nm^3$] | 6[2] | |
| NMHC | [kg/a] | 12 | 2 |
| | [mg/$Nm^3$] | 2[2] | 0.3 |
| $N_2O$ | [mg/$Nm^3$] | 0[1] | 0[1] |

1 No data available; zero emissions are assumed.

2 Only total hydrocarbon emission factor is given. A $CH_4$ : NMHC ratio of 3:1 as usual for heating systems is assumed here.

Source: EES, 1996, Knappstein, 1996, MTU, 1996, Fritsche *et al.*, 1995, Seier, 1996

### 5.3.3 Emissions

***CHP plants***

In contrast to conventional burning processes, the fuel cells operate on significantly lower reaction temperatures. That is the reason why fuel cell CHP plants have got lower specific emissions of pollutants as compared with equivalent conventional power plants. The specific $CO_2$ emissions are also comparatively low because of the high total efficiency.

**Table 5.22** Emission factors due to production of the most important construction materials (kg emission per t material produced; for $CO_2$: t emission per t material produced)

| | $SO_2$ | $NO_x$ | Staub | $CO_2$ | CO | $CH_4$ | NMHC | $N_2O$ |
|---|---|---|---|---|---|---|---|---|
| High-grade steel | 342 | 10 | 13 | 5 | 29 | 16 | 5 | 0.1 |
| Steel | 5 | 3 | 9 | 2 | 30 | 9 | 1 | 0.02 |
| Graphite[1] | 0.2 | 0.5 | 0.05 | 0.03 | 0.04 | 15 | 0 | 0.002 |
| Phosphoric acid | 20 | 2 | 2 | 1 | 0.5 | 2 | 2 | 0.03 |
| Platinum | 4e6 | 21e3 | 14e3 | 13e3 | 8e3 | 42e3 | 3e3 | 149 |
| Nickel | 2510 | 31 | 10 | 14 | 9 | 29 | 16 | 0.4 |

[1] Emission factor of coal

Source: Frischknecht *et al.*, 1996

In Table 5.21 the flue gas concentrations of the two fuel cell CHP plants are listed. The specific emissions of the pollutants $NO_x$, $SO_2$, particulates, $CH_4$, $N_2O$ and CO from fuel cell plants are significantly lower than those from conventional power plants whereas the $CO_2$-emissions mainly depend on the fuel used.

Additionally, some traces of acid, metal, and $NH_3$ are emitted from the PAFC. The emission of the electrolyte phosphoric acid of the PC 25 is in the range of 2.5–5 kg per year, the corrosion-caused emissions of iron is about 1 kg/year. The trace elements nickel and chrome are emitted with a rate of about 1.5–2 g/year. The $NH_3$ emissions, which are about 10–15 g/year, results from the ammonia content of natural gas (Ahn, 1996). For the MCFC-CHP plants, there exists no information about further emissions resulting from the operation process. However, due to the high content of nickel and steel in the MCFC CHP plant, traces of iron and nickel are probably emitted from those facilities, too.

***Emissions and other effects from the production of the fuel cells***

The emissions due to the production the fuel cell itself are quantified based on the emission factors for the production of the most important construction materials given by (Frischknecht *et al.*, 1996) (see Table 5.22). Most striking in the list of emission factors for the different construction materials are the high emission factors for the production of nickel and especially platinum. The production processes of these metals are very energy-intensive, especially the melting of the ore.

Based on these emission factors and the estimates of the construction materials employed in the production of the fuel cells, the emissions from the production of the fuel cells CHP plants have been calculated. The annualised results are given in Table 5.23 and Table 5.24. It is only a crude estimate because the composition of the fuel cells is only roughly estimated (see above).

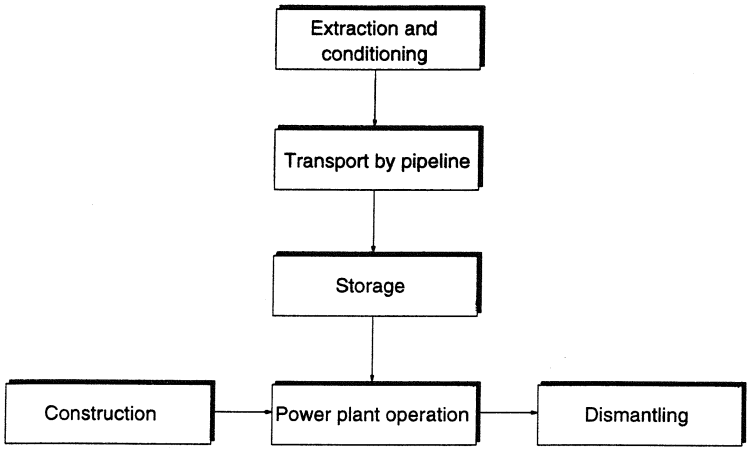



**Figure 5.14** Process steps of the gas-to-electricity fuel cycle

The gas fuel cycle is described for the gas-fired condensing boiler in section 3.1 (see Figure 5.13). The emissions are quantified accordingly. Again, direct emissions from gas extraction, conditioning and transport in the GUS have been quantified with the high leakage rates given in VDEW 1.0 as well as with the lower leakage given in (Frischknecht *et al.*, 1996). Especially the $SO_2$ and $NO_x$ emissions of the gas fuel cycle are significantly higher than in former EXTERNE analysis (e.g. (European Commission, 1995b)) because of the direct $SO_2$ and $NO_x$ emissions not taken into account before. Table 5.23 and Table 5.24 summarise the annual emissions for the different process steps.

For both fuel cell CHP plants, the production of the fuel cell contributes the most to the total $SO_2$ emissions. The contribution of the generation stage is small or even insignificant – with the exception of $CO_2$. Annual emissions are allocated to heat or to electricity production based on the exergy content of the co-products (see chapter 4)). Table 5.23 and Table 5.24 list the emissions allocated to heat and to electricity production. With the exception of the $SO_2$ and $NO_x$ emissions, the emissions allocated to electricity production are comparable to emissions from the conventional gas-to-electricity fuel cycle but lower than from other fossil fuel cycles. $SO_2$ emissions are much higher due to the high $SO_2$ emissions of the construction phase. If the construction emissions are not taken into account the $SO_2$ emissions allocated to electricity production are only about 140 mg/kWh for the PAFC plant and about 120 mg/kWh for the MCFC plant and thus comparable.

**Table 5.23** Annual emissions [kg/year] ($CO_2$ [t/year]) of the PAFC fuel cell CHP plant and its upstream processes

| | $SO_2$ | $NO_x$ | TSP | $CO_2$ | CO | $CH_4$ | NMHC | $N_2O$ |
|---|---|---|---|---|---|---|---|---|
| Gas extraction | 30 | 46 | 0.4 | 9 | 17 | 179<br>–1217 | *17*<br>*–88* | 0.3 |
| Gas conditioning | 136 | 16 | *25* | 6 | 8 | 317<br>–656 | 1.2<br>–1.7 | 0.2 |
| Gas transport | 0.1 | 56 | 0.7 | 10 | 27 | 485 | 4 | 0.4 |
| Total of the gas fuel cycle | 166 | *117* | 26 | 24 | *51* | *981–2359* | 22–93 | *0.9* |
| Construction[1] | *608* | 10 | 15 | 6 | 35 | 19 | 5 | 0.1 |
| Generation | 0 | 18 | 0 | *496* | 37 | 37 | 12 | 0 |
| Total | 775 | 146 | 41 | 526 | 123 | 1037-2415 | 39-110 | 1 |
| Emissions [kg/TJ] allocated to heat production (15.1%)[2] | 29 | 5 | 2 | 20 | 5 | 39-90 | 1-4 | 0.04 |
| Emissions [mg/kWh] allocated to electricity production (84.9%)[2] | 658 | 124 | 35 | 446 | 105 | 880-2050 | 33-93 | 0.9 |

[1] based on the projected lifetime (40 000 full load hours)

[2] based on exergy (flow temperature of 100°C, out-going temperature 40°C, ambient temperature of 15°C)

### 5.3.4 Priority impact pathways

For the two fuel cell reference energy systems the same impact pathways are important as for the other fossil energy systems (European Commission, 1995b):

- Effects of atmospheric pollution on
  - human health,
  - crops,
  - forests and unmanaged ecosystems,
  - materials
- occupational health effects, and
- impacts of global warming.

For the identification of other possible priority impact pathways, a closer look was taken at the production process of the fuel cell. The production process and the general characteristics of the most important construction materials of the fuel cell CHP plants (compare Table 5.22) are described in (Falbe, 1992). Steel, high-grade steel graphite and phosphoric acid are produced in large quantities. The production process of steel and of graphite are uncritical with regard to the environmental impacts as well as to the disposal. Phosphoric acid is produced by the reaction of sulphuric acid with phosphate rock and mainly used in the production of fertiliser. With regard to environmental effects, the production process of phosphoric acid is un-

**Table 5.24** Annual emissions [kg/year] ($CO_2$ [t/year]) of the MCFC fuel cell CHP plant and its upstream processes

| | $SO_2$ | $NO_x$ | TSP | $CO_2$ | CO | $CH_4$ | NMHC | $N_2O$ |
|---|---|---|---|---|---|---|---|---|
| Gas extraction | 28 | 42 | 0.4 | 8 | 15 | 164<br>*–1118* | *16*<br>*–81* | 0.2 |
| Gas conditioning | 125 | 14 | *23* | 5 | 7 | 291<br>–603 | 1.1<br>–1.5 | 0.2 |
| Gas transport | 0.1 | 52 | 0.7 | 9 | 24 | 446 | 4 | 0.4 |
| Total of the gas fuel cycle | 153 | *108* | 24 | 22 | *47* | *901–2166* | *20–86* | 0.8 |
| Construction[1] | *260* | 5 | 6 | 3 | 14 | 8 | 3 | 0 |
| Generation | 3 | 20 | 0 | *460* | 8 | 0 | 2 | 0 |
| Total | 415 | 133 | 30 | 484 | 69 | 909–2174 | 25–90 | 0.9 |
| Emissions [kg/TJ] allocated to heat production (6.3%)[2] | 14 | 4 | 1 | 16 | 2 | 30–72 | 0.8–3.0 | 0.03 |
| Emissions [mg/MWh] allocated to electricity production (93.7%)[2] | 314 | 101 | 22 | 366 | 52 | 687-1643 | 19-68 | 0.7 |

[1] based on projected lifetime (40 000 full load hours)

[2] based on exergy (flow temperature of 100°C, out-going temperature 40°C, ambient temperature of 15°C)

problematic, too. For the disposal of phosphoric acid, it must be purified and then precipitated with lime. The resulting sludge can be disposed in a secure landfill (Prager, 1995). Beside its use for catalytic processes, platinum is used for electric contacts, electrodes, etc. Platinum can be recycled although that process has not yet been tested on a larger scale (Scholta, 1994). With the exception of its high emissions (see above), platinum production does probably not have significant negative environmental aspects. Nickel is mainly used for alloys as catalyst or as electrode material. If inhaled, nickel has probably got carcinogenic effects (Koch, 1995), this aspect, however, is covered by occupational safety measures during the production process. After use nickel is usually recycled. In summary, it can be concluded that with the exception of the air-borne emission the production of the fuel cells do not cause significant environmental impacts and, thus, do not have to be considered as additional priority impact pathways in this analysis.

So far, possible environmental effects in case of a defect in a fuel cell have not been reported. However, fuel cells have not a long history of use, at least commercially. it can be speculated that (e.g.) in the case of a defect in a PAFC, acid used as an electrolyte in the stack could be emitted. Any subsequent effects would be extremely localised, and seem unlikely to generate a significant externality.

The electrochemical process does not produce sound. Hence, sound emissions of the plant originates from the additional units like pumps, ventilation and coolers only. In total the operation of a fuel cell CHP plant generates only low sound emissions. For instance, a value of 60 dB(A) at a distance of 10 m was measured at the selected reference CHP plant PC 25 (EES, 1996). This value is e.g. much lower than that for a single passenger car.

Since fuel cells are more efficient than Carnot-limited power plants, especially for operation in small units, they produce less waste heat. Due to the low thermal, visual and acoustic impacts of small type power plants, they can be built in areas with a high settlement density e. g. city centres.

### 5.3.5 The Quantification of Damage Costs

Contrarily to the conventional fossil fuel cycles, most of the $SO_2$, $NO_x$ and particulates emissions are due to the emissions of the gas fuel cycle and of the construction of the fuel cell. A site-specific approach is not possible here as specific sites cannot be quantified. About three quarters of the emissions of the gas fuel cycle occur in the GUS, while the construction materials for the fuel cells are produced to a high percentage in South Africa, other production sites are Canada, the GUS and Europe (Frischknecht *et al.*, 1996). As an auxiliary, the damage costs per tonne pollutant quantified by Krewitt and Heck (1997) are employed.

As site-specific ozone models are not available at the moment, ozone impacts are assessed using the damage costs derived by Rabl and Eyre (1997) with a simplified approach. In this approach results for the ozone models of EMEP and Harwell were combined with the exposure-response functions and economic valuation for health and agricultural crops recommended in (European Commission, 1995) to derive average damage factors for $NO_x$, NMHC and $CH_4$ emissions in Europe (incl. GUS).

Table 5.17 and Table 5.27 summarise the annual damage costs quantified for the two reference fuel cell CHP plants and their upstream processes. Mortality damages are only given employing the YOLL approach. Excluding global warming for the PAFC system the annual damages of the CHP plant itself of 250 ECU are negligible compared to the annual damages of the gas fuel cycle (ca. 2700 ECU) and of the construction of the CHP plant (5500 ECU). The respective results for the MCFC system are 340 ECU, ca. 2500 ECU and 2350 ECU, i.e. for the MCFC system the emissions of the upstream processes contribute several orders of magnitude more to the damages than the CHP plant.

However, for the global warming damages the opposite is true. The global warming damages of the CHP plants are by a factor 10 higher than those of the gas fuel cycle and by a factor 100 higher than those of the plant construction phase. Employing the EXTERNE results (illustrative restricted range) for global warming (see Appendix V) for the PAFC system annual damages of 9900–26500 ECU are calculated, i.e. damages about a factor 1-2 higher than the other damages together. The respective results for the MCFC system are 9100–24400 ECU.

**Table 5.25** Damages of the PAFC CHP plant and its upstream processes (negative numbers constitute benefits)

| | Annual damages [ECU/year] | Damages all. to heat production (15.1%)[1] [mECU/MJ] | Damages all. to elect. production (84.9%)[1] [mECU/kWh] |
|---|---|---|---|
| CHP PLANT | | | |
| *Public health* | | | |
| Mortality – YOLL[2] | 218 | 0.008 | 0.2 |
| of which TSP | 0 | 0 | 0 |
| $SO_2$ | 0 | 0 | 0 |
| NOx | 218 | 0.008 | 0.2 |
| CO | 0.02 | 0.000001 | 0.00002 |
| Morbidity | 24 | 0.0009 | 0.020 |
| Accidents | n.q. | n.q. | n.q. |
| *Occupational health (see below)* | | | |
| *Ozone impacts (human health & crops) – YOLL[2] (VSL)* | 43 | 0.002 | 0.04 |
| *Crops* | 0 | 0 | 0 |
| *Ecosystems* | i.q. | i.q. | i.q. |
| *Materials* | 2 | 0.00007 | 0.002 |
| Monuments | n.q. | n.q. | n.q. |
| *Noise* | ng | ng | ng |
| *Visual impacts* | ng | ng | ng |
| *Global warming*[3] | | | |
| Conservative 95% confidence interval | 1888-69052 | 0.07-2.6 | 1.6-58.6 |
| Illustrative restricted range | 8942-22852 | 0.33-0.9 | 7.6-19.4 |
| GAS FUEL CYCLE | | | |
| *Public health* | | | |
| Mortality – YOLL[2] | 1975 | 0.07 | 1.7 |
| of which TSP | 378 | 0.01 | 0.3 |
| $SO_2$ | 1214 | 0.05 | 1.0 |
| NOx | 382 | 0.01 | 0.3 |
| CO | 0.03 | 0.000001 | 0.00003 |
| Morbidity | 242 | 0.01 | 0.2 |
| Accidents | n.q. | n.q. | n.q. |
| *Occupational health* | 21 | 0.001 | 0.02 |
| *Ozone impacts (human health & crops) – YOLL[2] (VSL)* | 327–577 | 0.01–0.02 | 0.3–0.5 |
| *Crops* | 18 | 0.0007 | 0.02 |
| *Ecosystems* | i.q. | i.q. | i.q. |

| | Annual damages [ECU/year] | Damages all. to heat production (15.1%)[1] [mECU/MJ] | Damages all. to elect. production (84.9%)[1] [mECU/kWh] |
|---|---|---|---|
| *Materials* | 38 | 0.001 | 0.03 |
| Monuments | n.q. | n.q. | n.q. |
| *Noise* | ng | ng | ng |
| *Visual impacts* | ng | ng | ng |
| *Global warming*[3] | | | |
| Conservative 95% confidence interval | 281-10261 | 0.01-0.4 | 0.2-8.7 |
| Illustrative restricted range | 1329-3396 | 0.05-0.1 | 1.1-2.9 |
| CONSTRUCTION OF CHP PLANT | | | |
| *Public health* | | | |
| Mortality – YOLL[2] | 4699 | 0.2 | 4.0 |
| of which TSP | 218 | 0.01 | 0.2 |
| $SO_2$ | 4448 | 0.2 | 3.8 |
| NOx | 33 | 0.001 | 0.03 |
| CO | 0.02 | 0.000001 | 0.00002 |
| Morbidity | 573 | 0.02 | 0.5 |
| Accidents | n.q. | n.q. | n.q. |
| *Occupational health* | n.q. | n.q. | n.q. |
| *Ozone impacts (human health & crops) – YOLL[2] (VSL)* | 22 | 0.001 | 0.02 |
| *Crops* | 67 | 0.003 | 0.06 |
| *Ecosystems* | i.q. | i.q. | i.q. |
| *Materials* | 139 | 0.005 | 0.12 |
| Monuments | n.q. | n.q. | n.q. |
| *Noise* | ng | ng | ng |
| *Visual impacts* | ng | ng | ng |
| *Global warming*[3] | | | |
| Conservative 95% confidence interval | 24-894 | 0.001-0.03 | 0.02-0.8 |
| Illustrative restricted range | 116-296 | 0.004-0.01 | 0.1-0.3 |

[1] based on exergy (flow temperature of 100°C, out-going temperature 40°C, ambient temperature of 15°C)

[2] YOLL = mortality impacts based on 'years of life lost' approach; here results are calculated assuming a discount rate of 3 %.

[3] $CO_2$ equivalent emissions are calculated from the $CH_4$ and $N_2O$ emissions employing the Global Warming Potentials of IPCC (1996a) for the time horizon of 100 years.

ng: negligible; nq: not quantified; iq: only impact quantified; - : not relevant

**Table 5.26** Sub-total damages of the PAFC CHP plant and its upstream processes (mortality impacts valued with YOLL approach)

| Excl. construction (*incl. construction*) | Annual damages [ECU/year] | Damages all. to heat production (15.1%)[1] [mECU/MJ] | Damages all. to elect. production (84.9%)[1] [mECU/kWh] |
|---|---|---|---|
| Conservative 95% confidence interval | 4967-82472 *(10491-88865)* | 0.2-3.1 *(0.4-3.3)* | 4.2-70.0 *(8.9-75.4)* |
| Illustrative restricted range | 12658-29406 *(18274-35202)* | 0.5-1.1 *(0.7-1.3)* | 10.7-25.0 *(15.5-29.9)* |

[1] based on exergy (flow temperature of 100°C, out-going temperature 40°C, ambient temperature of 15°C)

Table 5.26 and Table 5.28 summarise the totals of the quantified damages, depending on the global warming damage costs per tonne $CO_2$ and whether the damages duie to the plant construction phase are considered. To the latter especially high uncertainties are attached. The annual damages are allocated to two co-products heat and electricity based on exergy (see chapter 4). Thus, allocated to electricity for the PAFC system excluding the construction phase total damages of 11–25 mECU/kWh are quantified (illustrative restricted range). The damages rise to 16–30 mECU/kWh if the construction phase is also taken into account. The respective results for the MCFC system are 9–21 mECU/kWh and 11–23 mECU/kWh. These results are comparable to those of the conventional gas-to-electricity fuel cycle (Krewitt *et al.*, 1997). That can be explained by the fact that the $CO_2$ emissions of a fuel cell plant are not lower than that of plants with a Carnot-limited plants and that in (Krewitt *et al.*, 1997) the direct emissions of $SO_2$ and $NO_x$ during gas extraction and conditioning are not taken into account.

Allocated to heat produced for the PAFC system including the construction phase total damages of 0.7–1.3 mECU/MJ are quantified. The respective results for the MCFC system are 0.5–1.0 mECU/MJ. These damage costs are about a factor 10 lower than that quantified for fossil heating systems in section 3.1.

**Table 5.27** Damages of the MCFC CHP plant and its upstream processes (negative numbers constitute benefits)

| | Annual damages [ECU/year] | Damages all. to heat production (6.3%)[1] [mECU/MJ] | Damages all. to elect. production (93.7%)[1] [mECU/kWh] |
|---|---|---|---|
| CHP PLANT | | | |
| *Public health* | | | |
| Mortality – YOLL[2] | 277 | 0.009 | 0.2 |
| of which TSP | 0 | 0 | 0 |
| $SO_2$ | 35 | 0.001 | 0.03 |
| NOx | 242 | 0.008 | 0.2 |
| CO | 0.01 | 0 | 0 |
| Morbidity | 30 | 0.001 | 0.02 |
| Accidents | nq | nq | nq |
| *Occupational health (see below)* | | | |
| *Ozone impacts (human health & crops) – YOLL[2] (VSL)* | 32 | 0.001 | 0.02 |
| *Crops* | -0.1 | 0 | 0 |
| *Ecosystems* | i.q. | i.q. | i.q. |
| *Materials* | 3 | 0.00001 | 0.002 |
| Monuments | nq | nq | nq |
| *Noise* | ng | ng | ng |
| *Visual impacts* | ng | ng | ng |
| *Global warming*[3] | | | |
| Conservative 95% confidence interval | 1748–63940 | 0.06-2.1 | 1.3-48.3 |
| Illustrative restricted range | 8942-22852 | 0.3-0.7 | 6.3-16.0 |
| GAS FUEL CYCLE | | | |
| *Public health* | | | |
| Mortality – YOLL[2] | 1821 | 0.06 | 1.4 |
| of which TSP | 349 | 0.01 | 0.3 |
| $SO_2$ | 1119 | 0.04 | 0.8 |
| NOx | 353 | 0.01 | 0.3 |
| CO | 0.03 | 0 | 0 |
| Morbidity | 224 | 0.01 | 0.2 |
| Accidents | nq | nq | n.q. |
| *Occupational health* | 21 | 0.001 | 0.02 |
| *Ozone impacts (human health & crops) – YOLL[2]* | 301–532 | 0.01–0.02 | 0.2–0.4 |
| *Crops* | 17 | 0.001 | 0.01 |
| *Ecosystems* | i.q. | i.q. | i.q. |

| | Annual damages [ECU/year] | Damages all. to heat production (6.3%)[1] [mECU/MJ] | Damages all. to elect. production (93.7%)[1] [mECU/kWh] |
|---|---|---|---|
| *Materials* | 35 | 0.002 | 0.03 |
| Monuments | nq | nq | nq |
| *Noise* | ng | ng | ng |
| *Visual impacts* | ng | ng | ng |
| *Global warming*[3] | | | |
| Conservative 95% confidence interval | 156-9415 | 0.01-0.3 | 0.1-7.1 |
| Illustrative restricted range | 741-3116 | 0.02-0.1 | 0.6-2.4 |
| CONSTRUCTION OF CHP PLANT | | | |
| *Public health* | | | |
| Mortality – YOLL[2] | 2006 | 0.1 | 1.5 |
| of which TSP | 87 | 0.003 | 0.1 |
| $SO_2$ | 1902 | 0.1 | 1.4 |
| NOx | 16 | 0.001 | 0.01 |
| CO | 0.01 | 0 | 0 |
| Morbidity | 245 | 0.01 | 0.2 |
| Accidents | nq | nq | nq |
| *Occupational health* | nq | nq | nq |
| *Ozone impacts (human health & crops) – YOLL*[2] | 11 | 0.0004 | 0.01 |
| *Crops* | 29 | 0.001 | 0.02 |
| *Ecosystems* | i.q. | i.q. | i.q. |
| *Materials* | 59 | 0.002 | 0.04 |
| Monuments | nq | nq | nq |
| *Noise* | ng | ng | ng |
| *Visual impacts* | ng | ng | ng |
| *Global warming*[3] | | | |
| Conservative 95% confidence interval | 24-894 | 0.0004-0.01 | 0.001-0.3 |
| Illustrative restricted range | 116-296 | 0.002-0.005 | 0.04-0.1 |

[1] based on exergy (flow temperature of 100°C, out-going temperature 40°C, ambient temperature of 15°C)

[2] YOLL = mortality impacts based on 'years of life lost' approach; here results are calculated assuming a discount rate of 3 %.

[3] $CO_2$ equivalent emissions are calculated from the $CH_4$ and $N_2O$ emissions employing the Global Warming Potentials of IPCC (1996a) for the time horizon of 100 years.

ng: negligible; nq: not quantified; iq: only impact quantified; - : not relevant

**Table 5.28** Sub-total damages of the MCFC CHP plant and its upstream processes (mortality impacts valued with YOLL approach)

| Excl. construction *(incl. construction)* | Annual damages [ECU/year] | Damages all. to heat production (6.3%)[1] [mECU/MJ] | Damages all. to elect. production (93.7%)[1] [mECU/kWh] |
|---|---|---|---|
| Conservative 95% | 4700-76300 | 0.2-2.5 | 3.5-57.7 |
| confidence interval | *(7000-79100)* | *(0.2-2.6)* | *(5.3-59.8)* |
| Illustrative restricted | 11800-27300 | 0.4-0.9 | 8.9-20.6 |
| range | *(14200-29800)* | *(0.5-1.0)* | *(10.7-22.5)* |

[1] based on exergy (flow temperature of 100°C, out-going temperature 40°C, ambient temperature of 15°C)

### 5.3.6 Excerpt on use of hydrogen as fuel (Hydrogen economy)

Fuel cells for transport could be run on a variety of fuels - e.g. methanol, natural gas, gasoline and hydrogen. Particular interest has focused on hydrogen and the development of a hydrogen economy, although hydrogen fuel cells are not necessarily the most promising technology for transport in the short term. Some experts judge that technical and economic obstacles will prevent hydrogen fuel cells from being widely used before the end of the first quarter of the next century (Shelef and Kukkonen, 1994) - with particular concern relating to the chance of developing a hydrogen economy. This section reviews some of the issues concerning the use of hydrogen as a transport fuel.

There are several studies on the technological aspects of a hydrogen economy, its gross energy requirement, its environmental burdens and even a preliminary study on the external costs available. Some results are given in the following, thereby focusing on fuel cell applications. However the presentation is not meant comprehensive.

An infrastructure to provide hydrogen would consist of the following components (Roesler, Zittel, 1994), (Wagner *et al.*, 1996):

*Hydrogen production from fossil sources*
Steam reforming of natural gas and partial oxidation of oil and coal are processes to produce hydrogen directly. Thereby, the whole carbon content of the fuel is converted to $CO_2$.. Hydrogen is also a by-product of some other processes, e.g. coal pyrolysis, oil reformation in raffineries, electric arc pyrolysis, fermentative processes.

*Hydrogen production from electrolysis*
Thereby, water is decomposed into the elements hydrogen and oxygen. For this process energy in the form of electricity is needed. Examples are the Alkaline Water Electrolysis (AWEL), the Solid Proton Conductor Electrolysis (SPEL) and the Solid Oxide Vapour Electrolysis (SOEL). The electricity needed for the process could be produced from fossil, nuclear and renewable sources. In most cases hydrogen production via electrolysis is not economical.

**Table 5.29** Primary energy requirement to provide 1 kWh of $GH_2$ or $LH_2$

| Hydrogen production cycle | Construction of facilities | Transport processes | Fuel | Total |
|---|---|---|---|---|
| *$GH_2$* | | | | |
| PV plant in northern Africa, AWAL | 0.997 | 0.015 | / | 1.012 |
| Solar Thermal Power Tower System in northern Africa, AWAL | 0.494 | 0.015 | / | 0.5 |
| Wind energy converter, AWAL | 0.251 | 0.005 | / | 0.256 |
| Gas, steam reforming | 0.022 | | 1.626 | 1.6 |
| Gas, KVAERNER process | 0.030 | | 3.139 | 3.2 |
| Biomass gasification | 0.018 | | 1.606 (0.159[1]) | 1.6 (0.18[1]) |
| Hydrogen as by-product of chemical processes | 0.012 | | 1.085[2] | 1.1 |
| *$LH_2$* | | | | |
| PV plant in Bavaria, AWAL | 2.819 | 0.03 | / | 2.8 |
| PV plant in northern Africa, AWAL | 1.744 | 0.058 | / | 1.8 |
| Solar Thermal Parabolic Trough plant, AWAL | 0.858 | 0.058 | / | 0.9 |
| Run-of-river power plant, AWAL | 0.094 | 0.030 | / | 0.1 |

[1] without calorific value of biomass
[2] calorific value of hydrogen produced as by-product

Source: Wagner *et al.*, 1996

*Hydrogen liquefaction*
On a commercial scale the Claude process is used to liquefy hydrogen.

*Hydrogen storage, transport and distribution*
Gaseous hydrogen ($GH_2$) can be transported in pipelines or steel cylinders. Hydrogen pipelines already exist, e.g. a pipeline in northern France, Belgium and the Netherlands, 330 km long, another in Houston, Texas, 64 km long. Long-distance pipelines should be underground with a cover of at least 1 m. Storage of gaseous hydrogen would be similar structured as the present storage of gas. Liquefied hydrogen ($LH_2$) can be transported by tankers and by trucks.

Environmental burdens from these processes are, in the first place, air-borne emissions, mostly due to the energy demand of the different processes. Wagner *et al.* (1996) have analysed the process steps of a hydrogen economy and quantified the gross energy requirements of several options, which give a first indication of the ranking of the different options with regard to the air-borne emissions (Table 5.29).

*Power generation and cogeneration from hydrogen with fuel cells*
PAFC, MCFC and SOFC CHP and power plants could be fuelled with $GH_2$ (see page 152). The plant itself would not emit any of the emissions under analysis here. The emissions due to the construction of the plant would not change relative to this analysis. Depending on the option to produce and supply the hydrogen the emissions of the upstream processes could significantly increase so that the net effect could be small.

*Application of fuel cells in transportation based on liquefied hydrogen*
Low-temperature fuel cells (AFC, SPFC) fuelled with $LH_2$ or compressed hydrogen could be employed in cars, buses and trains. Again, there are no air-borne emissions of the vehicles themselves, thus improving the air quality in urban areas. There is concern about the use of hydrogen in vehicles because of the high flammability and the low ignition energy of hydrogen. Experts maintain that hydrogen can be made as safe as gasoline or natural gas. the Daimler-Benz and BMW demonstration fleets, operated by trained personnel for several years in Germany, ran without mishap. However, the question remains how this experience can be extrapolated to self-serve stations for wide-spread public use (Shelef, Kukkonen, 1994).

Hörmandinger and Lucas (1996) have carried out a rough analysis of the environmental externalities of buses with fuel cells fuelled by hydrogen compared to diesel-driven buses. The hydrogen was produced in steam reforming. They employed two sets of externality adders per tonne pollutant ($SO_2$, $NO_x$, hydrocarbons, CO, particulates, $CO_2$) compiled from the literature. These externality adders are about one order of magnitude lower than the damage costs per tonne pollutant quantified in EXTERNE (see e.g. (Krewitt *et al.*, 1997)). For the fuel cell bus annualised social costs per vehicle of about 80000 US-$ were quanitified compared to 50000–90000 US-$ for the diesel bus.

### 5.3.7 Conclusions

Fuel cells have the potential to contribute in a significant way to emission reduction, especially of $CO_2$ and $NO_x$. Here, gas-fired fuel cell CHP plants were analysed. As first reference technology a PAFC CHP plant was defined because PAFC is the fuel cell with the highest stage of development. The second reference technology is a MCFC CHP plant. The fuel cell CHP plants emit practically no $SO_2$ and particulates and only less $NO_x$. The site of the two reference CHP plants is assumed to be Lauffen, 35 km north of Stuttgart.

The emissions of the upstream processes to provide the fuel natural gas were quantified based on the GEMIS 2.1 model employing the VDEW 1.0 database. Furthermore, the air-borne emissions due to the construction of the fuel stack were quantified based on a crude estimate of the materials used. Somewhat surprisingly, for both fuel cell CHP plants the production of the fuel cell contributes the most to the total $SO_2$ emissions. This is caused by the energy-intensive processes to produce platinum used in the PAFC plant and nickel used in the MCFC plant. In summary, contrarily to the conventional fossil fuel cycles, most of the $SO_2$, $NO_x$ and particulates emissions are due to the emissions of the gas fuel cycle and of the construction of the fuel cell.

The impacts and damages of the two reference energy systems were quantified following the EXTERNE methodology established by now. As with all the other fossil fuel cycles, increased mortality, mostly due to sulphate and nitrate aerosols, contributes the most to the total quantified damage costs (without global warming). With the exception of global warming the damage costs of the CHP plants themselves are negligible compared to the annual damages of the gas fuel cycle and of the fuel cell construction, which are a factor of 10–20 higher each. The present results for global warming are about a factor 1–2 higher than the other damages together. For global warming the CHP plants contribute about 90% of the total damages of the two reference energy systems.

Allocated according to the exergy of the co-products heat and electricity and using the so-called 'illustrative restricted range' EXTERNE results for global warming and the YOLL-approach for the valuation of mortality impacts, the following costs were quantified:

- PAFC system: 0.7–1.3 mECU/MJ heat produced; 16–30 mECU/kWh electricity produced;
- MCFC system: 0.5–1.0 mECU/MJ heat produced; 11–23 mECU/kWh electricity produced.

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## 5.4 Geothermal

***Presentation***

The main objective of the project is to analyse and assess the external costs and benefits of the geothermal-to-electricity fuel cycle in Portugal (São Miguel Island).

Several studies have been conducted on external costs in the framework of the called ExternE project both in the EU and in the USA. They encompass fossil fuels, nuclear, renewables and rational use of energy. The methodology defined has been applied to different cases (fuels, technologies and sites) but the geothermal-to-electricity fuel cycle has not been studied yet in Portugal nor in another country of the EU.

Renewables for energy generation are thought to be less environmentally harmful than fossil fuels, nevertheless any reliable economic calculations of environmental impacts exist that could answer with some confidence to this postulate. The main objective of this project is to assess the externalities (costs and benefits) generated by the production of electricity from a geothermal energy source.

***Geothermal resources: general outlook***

Geothermal energy is generally considered as a renewable energy source. Nevertheless, there is, as for biomass, a sustainability assumption which determine the exploitation level of the resource.

In reality, geothermal energy resources are made up of both non-renewable and renewable parts: the non-renewable is stored from the time of the earth's formation and the renewable one is due to the isotopic decay of radioactive elements.

A geothermal resource can be simply defined as a reservoir from which heat can be extracted and utilised for generating electric power or any other suitable industrial, agricultural or domestic application.

The geothermal systems that can at present be utilised - the *hydrothermal ones* - can be classified under three main types (Cesen Report 2; UNECE, 1985).

*Hot water systems*
These geothermal systems contain water at 50°C to 100°C and can be used for heating homes, and for various agricultural and industrial applications.

*Liquid-dominated systems*
These systems contain pressurised water, at a temperature considerably higher than 100°C, and limited quantities of stream. Being covered by a impermeable layer, the fluid cannot escape to the surface and is kept under pressure. These are the geothermal systems more usually suited for industrial exploitation: electric power generation is economically convenient, while

there can be other uses for the hot waters. When a well reaches a reservoir of this type, pressurised water rises up in the well, with the consequence of a rapid fall in pressure but not a similarly rapid drop in temperature. Part of the water vaporises and, as a final result, the well produces water and stream, with water usually being predominant.

*Vapour-dominated systems*
There is a geological similarity to the previous ones: in them water and steam co-exist, the latter as a continuous and prevalent phase. This type of system produces dry, generally superheated steam, with small quantities of other gases. The typical utilisation of superheated steam is electricity generation.

*Other geothermal systems*
Other geothermal systems not so far exploited commercially are:

- hot dry rocks, requiring the creation of an artificial geothermal reservoir;
- geopressurised systems, in which water is found at a temperature between 150 and 200°C or more (with presence of methane) trapped in sedimentary rocks. It is not certain that from such systems it may be possible to extract energy (kinetic or thermal, because of the presence of steam and methane), due to considerable technological problems and high exploitation costs.

Most of geothermal resources are hydrothermal. Geopressured magma and hot dry rock systems are in the demonstration stage (Palmerini, 1993).

*Methodological approach*

The methodological approach will follow the one undertaken in the framework of the *ExternE* project and already applied to several fuel cycles in Europe and USA. This methodology has been described extensively in previous works (European Commission, 1995a).

The methodological concepts developed in the ExternE project will be implemented in one case in order to quantify external costs and benefits, to test the methodology with a new fuel cycle and to make some recommendations on areas in which further research is required.

### 5.4.1 Reference site, technology and emissions

In this section is made a brief description of the site and the technology of the reference geothermal power plant.

***Reference site***

As shown in Figure 5.15 and Figure 5.16 (see hereafter) the geothermal power plant is located in São Miguel island in the North slope of Água de Pau, between Cachaços and Lombadas sites, municipality of Ribeira Grande, district of Ponta Delgada. São Miguel is the larger island (with an area of about 747 $km^2$) of the Azores Archipelago (North Atlantic ocean), a volcanic formation composed by nine islands.

**Table 5.30** Seismic activity of Fogo volcano

| Year | Recorded events | Localised events | % | ways/ /maximum intensity |
|---|---|---|---|---|
| 1986 (a) | 257 | 93 | 36 | 5/IV |
| 1987 (a) | 233 | 50 | 21 | 2/IV |
| 1988 (a) | 1 224 | 314 | 26 | 33/VI-VII |
| 1989 (b) | 217 | 101 | 47 | 2/II |
| 1989 (c) | 9 230 | 956 | - | 109/II |
| 1990 (b) | 140 (d) | 66 (e) | 47 | 1/II |
| Total | 11 044 | 1 589 | - | 152/II-VII |

(a) 2nd semester.
(b) 1st semester.
(c) Geothermal crises of July-September/89 + 4th trimester.
(d) Under analysis.
(e) More than 2 500 events recorded.

*Geomorphology and physiography*
S. Miguel island relief is characterised by alternated high and low points, which demonstrates its geological origin. Plateaus in the central part of the island make the exception to the general relief. The Maximum altitude is 1105 m in Pico da Vara located in the oriental side of the island. There are many volcanic cones, and many of them with craters (Sete Cidades, Furnas and Fogo). The geothermal plant is constructed in the north slope of Fogo volcano at an altitude of 525 m.

*Geology, volcanism and seismicity*
Fogo volcano is one of four active volcanoes in S. Miguel island. All recent scientific works about Fogo volcanism agree in one main point: this volcano is very explosive and undoubted the more dangerous of Azores islands (Booth *et al.* 1983; Forjaz, 1983 *fide* UNINOVA, 1991).

Besides the high degree of volcanism revealed, Fogo volcano is characterised by a significant micro and macro-seismicity. The selection of five representative years shows the intense seismic activity of the volcano (see Table 5.30 hereafter).

*Land use*
Near the geothermal plant, the area is characterised by different occupations: (i) agricultural; (ii) social areas; (iii) pasture; (iv) forestry; (v) natural vegetation; (vi) "cerrado" and orchards; The area affected to the different land uses is approximately estimated in Table 5.31. As it can be seen pasture is the major land use, followed by forestry and natural vegetation. This kind of occupation shows the very low agricultural value of soils surrounding the geothermal plant. The high slope and the abundant rainfall contribute to destroy soil structure.

**Table 5.31** Land use in the area

| Land use | Area ($km^2$) | % total area |
|---|---|---|
| Agriculture | 7.5 | 9 |
| Social areas | 7.0 | 9 |
| Pasture | 30.0 | 37 |
| Forestry | 19.0 | 24 |
| Natural vegetation | 15.0 | 19 |
| "Cerrado" and orchards | 1.5 | 2 |
| Total | 80.0 | 100 |

*Climate and hydrology*
The Climate in S. Miguel, like in the other islands of the archipelago is temperate, rainy, typically oceanic and varies greatly with the relief. The annual average temperature could vary from 17-18°C (Ponta Delgada) to 10-11°C (above 400-500 m). In the wind regime there is a predominance of strong winds from north-west direction, which associated with the type of relief allow to high atmospheric turbulence conditions.

Annual rainfall at the reference site reaches one of the highest values of the island, with about 3000 mm/year in the north slope of Água de Pau and 1500 mm/year in the coastal zone of Ribeira Grande. Thus the estimated value for average annual rainfall is between 2000 and 2500 mm/year.

Due to this type of climate the area is marked by a very significant hydrological surplus. Additionally to high superficial flow, there are many springs (some of them of hot water) distributed along the riverside Ribeira Grande. Associated to the springs there are aquifers with a high degree of permeability due to geological material (piroclastic). This characteristic could explain the contamination of water streams when the first geothermal well was constructed in 1988.

The water from springs present good quality and is directly used for domestic supply. The surface waters have also good quality and some times are used in agriculture. The municipality of Ribeira Grande is supplied by seven springs, two of them located in the area of the geothermal plant.

*Socio-economic and energetic characterisation*
S. Miguel island has a population of 125,915 inhabitants (INE, 1993) with 49% living in Ponta Delgada. The population density is of 169 inhabitants per $km^2$. The municipality of Ribeira Grande has a population of 27,163 inhabitants. Across decades agriculture and cattle breeding represented the key-sector in the region. The economic system has a significant fragility and external dependency.

The electric generation sector in S. Miguel is characterised by two hydropower plants, three thermoelectric power plants and two geothermal power plants (one already installed with a power output of 3 MW and the other one in conclusion, the reference plant, with 12.7 MW). As it can be seen in Table 5.32 the electricity generation structure of the island is dominated

**Table 5.32** Electricity generation structure for S. Miguel island (1994)

| Type | MWh | % |
|---|---|---|
| Thermal | 145 646 | 74.5 |
| - Fuel | 145 353 | 99.8 |
| - Diesel | 292 | 0.2 |
| Hydropower | 16 462 | 8.4 |
| Geothermal | 33 440 | 17.1 |
| Total | 195 548 | 100 |

Source: Rosa Nunes, 1995.

**Table 5.33** CL-1 well data

| | |
|---|---|
| Well enthalpy | 275 kcal / kg |
| Well total flow | 125 t / h at well head pressure of 6 bar |
| Fluid temperature | 156.9 °C |
| Brine | 75.25% by weight of total flow |
| Steam | 23% by weight of total flow |
| Non-condensable gases | 1.75% by weight of total flow |

by imported energy (fuel oil), about 74.5%. The other types, hydropower and geothermal respectively, represent in 1994 about 8.4% and 17.1% of the total electricity generated.

***Reference technology***

*General presentation*

The reference plant has a gross power output of 12.66 MW and a net power output of 10.9 MW. At full capacity, the plant has a net electrical production of about 80.64 GWh/year, which represents about half of the total electricity consumption of the island in 1994.

The main technological modules are: (i) the plant with 5 energy converters, (ii) 4 geothermal wells with about 1 500 m depth each and complementary units (pipelines, steam separator and collector) and (iii) energy transmission network. One of wells, named CL-1 (2 029 m depth) was the first to be constructed and tested. The installation has two platforms, A and B, the first one for the well CL-1 and the second one for the central unit and remaining wells.

During exploration the unit receives at well's head about 500 t/h of geothermal fluid at a temperature of about 152° C, distributed between liquid (372 t/h of brine), steam (119 t/h) and non condensable gases (9 t/h).

Technical data are only available for the well CL-1 (see Table 5.33). The calculations assumed for the reference plant and for the 3 other wells (CL-2, CL-3 and CL-4) were derived from well CL-1.

**Table 5.34** Main characteristics of the reference case

| Characteristics | Values |
|---|---|
| Gross electricity generation (MW) | 12.66 |
| Electricity sent out (MW) | 10.83 |
| Operating time (h/year) | 7 446 |
| Life time of the power plant (years) | 20 |
| Life time of wells (years) | 10 |
| Number of geothermal wells | 4 |
| Energy content of the geothermal fluid (kcal/kg) | 275 (or 1.15 MJ) |
| Total wells flow (t/h) | 500 |
| Number of energy convertors | 5 |

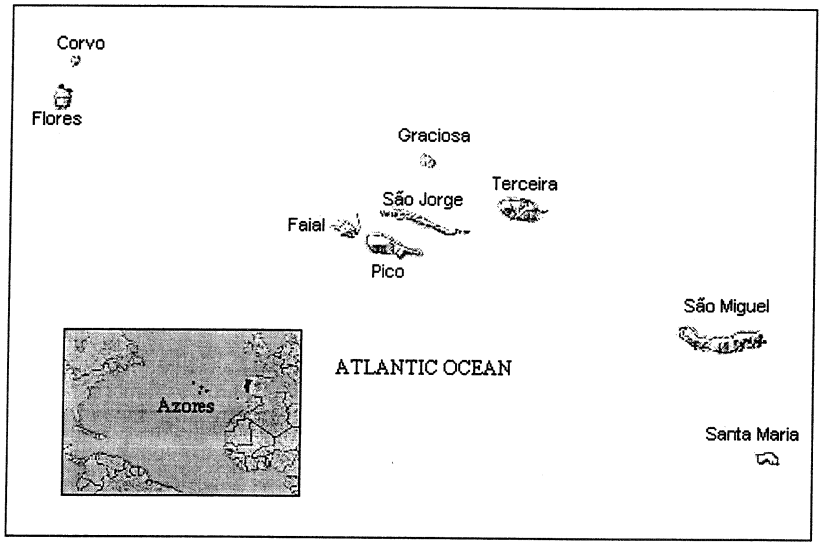



**Figure 5.15** Map of Azores islands

The main characteristics of the project can be summarised as shown in Table 5.34.

*Process description*
The geothermal fluid (see Figure 5.17) is led from the well head to a flash separator which separates the liquid (brine) from the saturated steam and non condensable gases (NCG). The steam flows from the separator to a vaporiser where the motive fluid is vaporised. All NCG and a small amount of steam are released to the atmosphere. The condensate is mixed with the brine at the inlet of the preheater. The brine is led from the separator via an accumulating vessel to the preheater and then discharged or reinjected in a well.

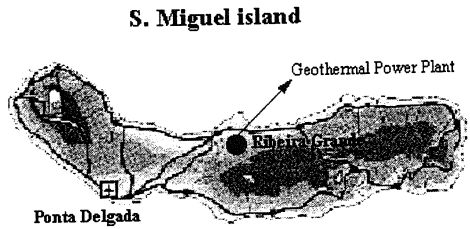



**Figure 5.16** Location of the reference plant in São Miguel island

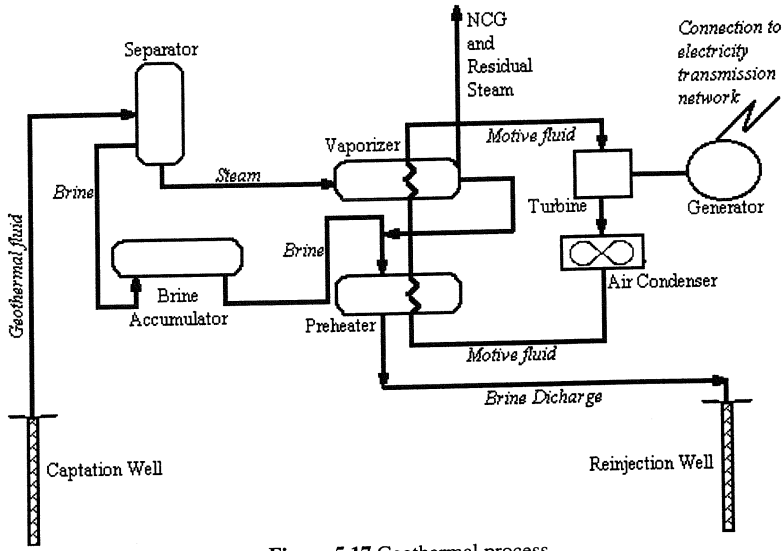



**Figure 5.17** Geothermal process

The system uses an organic working fluid (n-pentane) which is led to the preheater and from there to the vaporiser. From the vaporiser it is led to the turbine which is coupled to a generator. After expanding in the turbines, it flows to the air cooled condenser and from there via a feed pump back to the preheater.

***Emissions***

The analysis of the geothermal fuel cycle has to cover all the stages of the energy production process including drilling, construction and operation of the power plant including fluid reinjection and dismantling of the power unit.

*Well drilling and power plant construction*
Well CL-1 was drilled between November 1988 and February 1989. The original plan was to drill a deviated well. However, considerable drilling problems were encountered which have forced the well to be completed vertically. The well was drilled with mud and water. This approach coupled with the difficulty of servicing and supplying the operation, particularly after problems were encountered, was the source of the majority of the drilling problems.

The original well design of CL-1 was a deviated well of 2280 meter depth. The maximum deviation angle was 20° and the deviation starting point (kick off point) was planned to be at a depth of 600 meters. Well CL-1 has been completed as a vertical well to a depth of 2029 m.

After the completion of each one of the wells, a "short term test" was performed. The test took place after allowing the well to flow for an appropriate period of time for cleaning purposes. During the test some parameters were measured: (i) flow rate of steam versus well pressure; (ii) flow rate of water versus pressure; (iii) non-condensable gases quantity in the fluid at various well head pressure; (iv) chemistry of fluids; (v) down hole pressure at various flow rates of geothermal fluids.

The drilling of well CL-1 and the following tests have generated a significant amount of emissions namely the release of geothermal fluid and mud. Air emission during this phase are mainly due to fuel combustion and losses of geothermal fluid (saturated steam and NCG). These emissions have not been quantified but the characteristics of the fluid have been measured (see Table 5.35).

The power plant integrates two platforms established in an area of 2 ha where pasture was the dominant use. The first platform includes the well CL-1 and the second one comprises the other wells and the power station. The construction of these platforms has induced the removal of soil in an area dominated by pasture.

*Energy production, maintenance and dismantling*
Due to the geothermal fluid characteristics the wells could block up - scaling - with silica and calcite. This problem will slow down the flow of geothermal fluid that arrives to the well head. Therefore the wells must be periodically clean through mechanic operations during periods between 36 and 72 hours, with interruption of the cycle of electricity production. However, giving the structure of the geothermal plant modules, the unit could have a continual production, despite the maintenance and repairing operations.

After energy recovery, the liquid part of the geothermal fluid (487 t/h at a temperature of 90°C) could be discharged to the ocean directly or after treatment or pumped to a reinjection

**Table 5.35** Characteristics of reference geothermal fluid

| Chemical species | Minimum (mg/l) | Average (mg/l) | Maximum (mg/l) |
|---|---|---|---|
| $Ca^{2+}$ | 0.52 | 0.59 | 0.63 |
| $Mg^{2+}$ | <0.01 | <0.01 | <0.01 |
| $Na^{+}$ | 1 538 | 1 628 | 1 786 |
| $K^{+}$ | 224 | 237 | 260 |
| $Li^{+}$ | 0.85 | 0.90 | 0.99 |
| As | <0.49 | 0.60 | 0.93 |
| Fe | 0.03 | 0.11 | 0.16 |
| $Sr^{2+}$ | 0.07 | 0.08 | 0.08 |
| $H_2S$ | 3.48 | 3.68 | 3.88 |
| $NH^{4+}$ | 0.32 | 0.36 | 0.40 |
| $HCO^{3-}$ | 1 020 | 1 043 | 1 085 |
| $CO_3^{2-}$ | 14.6 | 18.5 | 40.8 |
| $SO_4^{2-}$ | 119 | 125 | 136 |
| $Cl^{-}$ | 1 890 | 1 989 | 2 172 |
| $F^{-}$ | 37.5 | 39.5 | 41.2 |
| $B^{-}$ | 3.79 | 3.98 | 4.33 |
| $SiO_2$ | 428 | 451 | 490 |
| pH | 7.45 | 8.07 | 8.90 |
| Total dissolved solids | 4 724 | 4 990 | 5 458 |

well. The last solution is under study considering the technical requirements and the economic impacts.

At the same time 4.3 t/h of steam and 8.75 t/h of NCG, mainly $CO_2$ (99.5%), $N_2$ (0.2%) and $H_2S$ (0.1%), are released to the atmosphere. Table 5.36 summarises the main emissions from power plant operation.

Giving the lifetime of the plant the dismantling phase may happen in 20 years, but this value is uncertain and is likely to be greater. The dismantling of the geothermal plant will probably be determined by the low production of the wells and not because of the deterioration of the equipment. The technical lifetime of the wells is estimated to be about 10 years. So the operation of the power plant during its lifetime will require the drilling of new wells within the area of the geothermal unit.

If the wells production becomes low, with the risk of slow down production, there is the possibility of drill new wells near the actual area or move the plant to another site.

**Table 5.36** Emissions from electricity generation stage

| | Atmospheric emissions (g/kWh output) | Liquid emissions (kg/kWh output) |
|---|---|---|
| NCG: | | |
| - $CO_2$ | 803.9* | - |
| - $H_2S$ | 0.808 | - |
| - $N_2$ | 1.616 | - |
| - Other gases | 1.616 | - |
| Motive fluid losses (n-pentane) | 0.002 | - |
| Residual steam | 367.0 | - |
| Brine (discharged) | - | 44.96 |

* Emission factor of $CO_2$ for a fuel oil plant with the same capacity and existing technology is about 676.8 g/kWh output.

**Table 5.37** Overview of major environmental impacts of geothermal energy developments

| Source(s) | Impact(s) | Endpoint(s) |
|---|---|---|
| | Development & Construction phase | |
| Presence and operation of drilling equipment | Aesthetic impacts (visual intrusion) | Visual amenity loss |
| | Noise impacts | Local amenity loss |
| | Air pollution impacts | Human health |
| | Water pollution | Water supply |
| | Ecological disturbances | Ecological effects |
| | Accidents | Local economy |
| | Employment | |
| Construction of roads and movement of heavy equipment | Noise impacts | Local amenity loss |
| | Accidents | Human health |
| | Landscape disruption | Ecological effects |
| | Disturbances to local flora and fauna | Road conservation |
| | Road damages | Local economy |
| | Employment | |
| Disposal and handling of drilling mud | Aesthetic impacts | Visual amenity loss |
| | Soil & water contamination | Soil & water treatment |
| | | Human health |

### 5.4.2 General outlook of impacts

A first overview on the environmental impacts related with the exploitation of geothermal energy sources was presented by Armstead (1976). The most important environmental endpoints of geothermal fuel cycle are basically dependent on (i) the type of geothermal unit under analysis (e.g. technological features, temperature of geothermal fluid, utilisation(s) of thermal energy) and on (ii) the stage of the cycle. The major stages of the fuel cycle that should be addressed are:

**Table 5.37** continued

| Source(s) | Impact(s) | Endpoint(s) |
|---|---|---|
| | Operation phase | |
| Operation of a geothermal unit in a site with high seismic activity | Risk of a well overthrow during a seismic event with water contamination | Water supply<br>Human health<br>Ecological effects upon vegetation |
| Release of non-condensable gases into the atmosphere | Air pollution ($CO_2$ & $H_2S$) | Global warming<br>Human health<br>Ecological effects upon vegetation |
| Liquid release into streams (condensates and brines which might contain toxic and corrosive substances) | Soil & water contamination | Water supply<br>Human health<br>Ecological effects |
| Thermal discharges into natural water bodies | Physical pollution of water | Ecological effects |
| Withdrawal of underground fluid | Subsidence | Property damages |
| Removal of geothermal fluid and pressurised reinjection of condensate or other spent fluids | Induced seismicity. Seismic effects include production of microseismic events | Property damages |
| Presence and operation of a geothermal unit | Aesthetic impacts<br>Employment<br>Recreational impacts | Change in visual amenity<br>Local economy |
| Pressurised steam flow | Noise | Occupational health<br>Local amenity loss |
| | Dismantling phase | |
| Non-adequate sealing of wells | Risk of a well overthrow with water contamination | Water supply<br>Human health<br>Ecological effects upon vegetation |
| Removal of constructed buildings and power plant facilities | Aesthetic impacts<br>Noise impacts<br>Soil & water contamination | Local amenity loss<br>Human health<br>Water supply |

- Heat mining (including the prospecting of the geothermal resource);
- Construction of the geothermal unit and general facilities (including possible distribution networks);
- Operation of geothermal unit;
- Dismantling.

The drilling of the wells, the testing of the productive ones and the construction of energy distribution and conversion systems are examples of activities which can disturb the environment, scatter dust, cause noise and aesthetic disturbance and so on.

Table 5.37 presents an overview of major environmental impacts from geothermal developments, including those that are local, site and fuel-cycle specific, and global.

The following impacts are considered for prioritary analysis:
- Global warming impacts due to greenhouse gas emissions;
- Potential impacts due to the locations of a geothermal plant in a zone of seismic and volcanic activity;
- Impacts on water quality from heat mining;
- Environmental impacts from geothermal fluid release;
- Impacts from induced seismicity or subsidence on infrastructures;
- Impacts of air pollutants on human health and terrestrial ecosystem.

A review of some other impacts is presented in section 5.4.10 although they are not assessed. Some of these impacts (aesthetic impacts and recreational benefits) can be important or trivial but at this point of the project this is not clear. For noise, the impacts could be significant if equipment to reduce emission level are not installed. But the noise generated during the operation of the power plant is thought to be acceptable. Impacts on employment are not considered in the framework of the ExternE project but their potential positive externalities should be estimated.

### 5.4.3 Global warming impacts due to greenhouse gas (GHG) emissions

***Introduction***

A great number of gases in the earth's atmosphere let solar radiation (visible light) pass to the surface of the earth while trapping infrared radiation (IR), also referred to as heat radiation, that is re-emitted by the surface of the earth. This heat radiation would have otherwise escaped to space. It is this trapping of infrared radiation that is known as greenhouse effect.

At a global average surface temperature of 288 K (15°C), the long-wave outgoing radiation from the surface of earth is 390 $Wm^{-2}$, compared to 236 $Wm^{-2}$ from the top layer of the atmosphere. This reduction in the long-wave emission is a measure of the greenhouse effect (Ramanathan, 1987).

Fuel cycles present in general more than one stage giving rise to emissions of greenhouse gases (GHG). These gases include mainly carbon dioxide ($CO_2$), methane ($CH_4$), nitrous oxide ($N_2O$), and trace amounts of other gases such as chlorofluorcarbons (CFC's). Geothermal fuel cycle is an exception to the above mentioned rule, since GHG emissions are almost entirely concentrated into a single stage: electricity generation.

**Table 5.38** $CO_2$ equivalents of the various greenhouse gases

| Unit mass (1g) of greenhouse gas | $CO_2$ equivalents (g) |
|---|---|
| $CO_2$ | 1 |
| $CH_4$ | 68.8 - 88.0 |
| | 58* |
| $N_2O$ | 150 - 250 |
| | 206* |
| CFC-11 | n.a. |
| CFC12 | n.a. |
| $O_3$ | 1833.3 |

* Values suggested by IPCC (1990).
n.a.: not available.

The most important greenhouse gas associated with geothermal activity is $CO_2$, although some other gases can be emitted in trace quantities in some stages of the fuel cycle, and in some particular geological and tectonic settings.

Table 5.38 gives the $CO_2$ equivalents, in a mass basis, of the different greenhouse gases.

***Origin and quantification of the emissions***

The flux of $CO_2$ from deep geothermal structures has been associated with the metamorphism of underground carbonate reservoirs, and namely with the metamorphism of marine carbonate rocks (Chiodini *et al.*, 1995). During the last few years some authors have compared the emissions from geothermal energy systems with other energy alternatives, and namely with fossil-based power plants. Goddard *et al.* (1990) provide proof of the reduced $CO_2$ emissions from geothermal over other energy alternatives. According to de Jesus (1995), fossil-based power plants are estimated to emit 20-25 times more $CO_2$ than geothermal plants. These estimates are in accordance with the results obtained by Barbier (1991) and Armannsson and Kristmannsdottir (1992), who did comparisons between $CO_2$ emissions at geothermal and fossil fuel-fired power plants (see Figure 5.18 hereafter).

The comparison carried out by Fridleifsson and Freeston (1994) was not clear in what concerns the stages considered for fossil fuel cycles. We assume that the values presented by the authors for those fuels take into account the emissions occurring in other stages such as mining and transportation. Furthermore, in order to include some sense of deviation, we took some further results achieved by the ExternE project and put them into the author's figure.

In general, as already pointed out, geothermal energy for electricity production is harnessed in the immediate vicinity of the resource, an thus the entire fuel cycle, from resource extraction to electric transmission, is generally located at one site. Unlike fossil fired stations, for which sometimes large emissions are expected during stages such as mining, transportation and waste disposal, GHG emissions from the geothermal fuel cycle under study are dominant at the electricity generation stage.

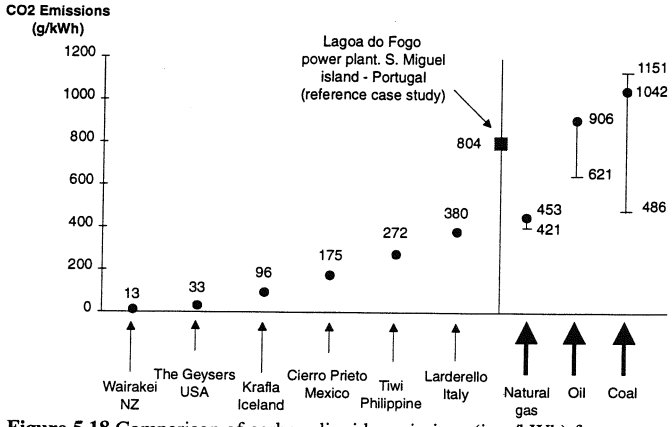



**Figure 5.18** Comparison of carbon dioxide emissions (in g/kWh) from geothermal and fossil-fired power plants (Sources: Fridleifsson and Freeston, 1994; EC, 1995b-c; CEEETA, 1995).

With respect to other renewables the comparison is not straightforward. Biomass and biogas, for instance, are not good examples for comparison since the carbon emitted during fuel combustion will equal the amount of carbon organically bound during plant growth (providing that the process is conservative, *i.e.*, plantation rate equals harvesting rate for a human time scale). The same happens with biogas, but at levels of the food chain ecologically more or less distant from primary production.

As far as GHG emissions is concerned, the comparison of geothermal with other renewable energy sources can be made for instance taken the cases of wind and hydro. For the case of wind, the value suggested in the ExternE project (European Commission, 1995d) is 9.1 g $CO_2$/kWh.

By looking at the figures drawn above it is noticeable the discrepancy between the $CO_2$ emissions of our case study and the average emissions per kWh observed in other geothermal developments. The maximum emissions per kWh were recorded in Italy, at the Larderello geothermal plant, and they correspond just to 380 g/kWh (47% of the emissions at Lagoa do Fogo power plant). By giving faith to the results available, presented in the EIA study of the project (UNINOVA, 1991), two possibilities remain for the explanation of this discrepancy: (i) the low efficiency of the power plant and/or (ii) the high rate of metamorphism of the subterranean carbonate reservoirs. The last one seems to play an important role in this case.

**Table 5.39** Total estimated damages from green house gases in the geothermal fuel cycle using a conservative 95% confidence interval (3.8-139 ECU/t of $CO_2$).

| | Damage costs (mECU/kWh) | |
|---|---|---|
| | Low | High |
| Power generation stage | 3.05 | 111.7 |
| Other stages | nq | nq |
| Total estimated cost | 3.05 | 111.7 |

nq: not quantified.

**Table 5.40** Total estimated damages from green house gases in the geothermal fuel cycle using an illustrative restricted range (18-46 ECU/t of $CO_2$).

| | Damage costs (mECU/kWh) | |
|---|---|---|
| | Low | High |
| Power generation stage | 14.47 | 37.0 |
| Other stages | nq | nq |
| Total estimated cost | 14.47 | 37.0 |

nq: not quantified.

***Monetary valuation of global warming***

The impacts of global warming are diverse and potentially problematic to evaluate from the economic point of view. Several thousands of good reference papers have been produced on the subject, but there seems to remain a great variability and uncertainty on the results achieved by the different authors. This is due to the valuation methodology used and also to the scenarios considered. The damages presented in the ExternE reports are based on estimates reported in the literature and were considered suitable for ExternE accounting framework.

The electricity generation stage is the most important stage regarding GHG emissions within this fuel cycle. Emissions from the other stages are negligible. Giving the damage cost established in the framework of this project (3.8-139 ECU per t of $CO_2$ equivalent, low and high estimates) and the GHG emissions estimated in our case, impacts from global warming are presented in Table 5.39 and Table 5.40 for two intervals of confidence.

### 5.4.4 Potential impacts due to the location of the geothermal wells in a zone of high seismic and volcanic activity

***Introduction***

The decision of installing a geothermal development in a zone of high seismic and volcanic activity must be done taking into account the operation risks involved. The possible rupture of wells and/or the damages that may be caused to the facility during a seismic event are consid-

ered as internalised: the plant owner has incorporated such risk into the economic evaluation of the project, and has safeguarded himself through, for instance, some scheme of insurance.

The externalities that might be expected in case of a seismic or volcanic event regard the possible damages caused by the presence of a fractured well that may allow the passage of geothermal fluid to the upper aquifers. This may result in the contamination of such aquifers, making its water improper for drinking or for other valuable purposes.

***Quantification of the emission: the record of seismic and volcanic events in São Miguel island***

The volcanic history of São Miguel has been studied in great detail and can be considered to be well known (Machado, 1967; Walker and Croasdale, 1971; Booth *et al.*, 1978; Booth *et al.*, 1983). Its seismic history is less well known, although a great amount of seismic data has been collected during the last several years. A study of seismicity of the central region of São Miguel made in 1983 (Dawson *et al.*, 1985; 1986) indicated continuous seismicity with frequent earthquake swarms on the east flank of Água de Pau volcano while the west flank of the volcano appeared to be aseismic.

The record of seismic events registered by the seismic station of IM between 1943 and 1968 shows a list of the number of felt earthquakes in Água de Pau area, which also included the highest intensity reported for each earthquake swarm. It includes 9 swarm periods of seismic activity with a total of 129 felt earthquakes. On four occasions during these 26 years, earthquakes of intensity VII (Mercalli) were reported. Thus for this period we have a frequency of about 0.15 seismic events/year with such an intensity.

Modern scales of seismic intensity (Richter scale) are based on magnitude. An earthquake of magnitude 0 can only be detected in the vicinity of its epicentre by very sensitive instruments, such as seismographs; of magnitude 3 it can be detected in a circumscribed area; of magnitude 4.5 it can cause some slight damages. Great seismic events have a magnitude of 7 and the most destructive ones recorded up to now have a magnitude of 8.6.

The intensity of earthquakes are also often described using the international scale (modified version of the former Mercalli scale), which comprises 12 degrees established according to a standard questionnaire based on the effects over populations and on the observed damages.

***Physical quantification of the impact***

The probability of a seismic event stronger enough to cause the rupture of an existing geothermal well will depend basically on the geological characteristics of the crust (*e.g.* tectonics and rock nature), and on the technical characteristics of the well. In general we may say that the geological structure's response to a given seismic event, as well as the well's one, are rather unpredictable. This is mainly because of the uncertainty about the relative position of the different geologic layers, because of the random nature of the seismic event itself and be-

cause of possible cumulative effects in the past (*e.g.* strain to which well was submitted during former seismic events).

In the last 53 years no seismic events with intensities over VII were observed in São Miguel. However, seismic events with intensities VI-VII are frequent in the region (about one per year, considering the seismic data available in Forjaz *et al.*, 1991). Machado (1967), in his review on the seismic and volcanic history of S. Miguel, refers the earthquake of October 22th of 1522, that in some areas of the island (e.g. Vila Franca) has reached the magnitude X (Mercalli).

Walker (1971) has found that at least must there have been 6 extremely destructive plinian periods in S. Miguel in the last 5000 years. Further, Both *et al.* (1983) and Forjaz (1985) have concluded that the plinian periods should be expected each 150 years (Both) or 450 years (Forjaz).

If we assume, as an hypothetical figure, that the annual probability of a seismic event capable to cause the rupture of an existing well is about 0.01 (*i.e.*, a well will be destroyed after 100 years with probability 1), then this could be the basis for the physical quantification of this impact. However, we should keep in mind that after, say, 30 years the well will be clogged up and thus no impacts are expected after that. But besides this number, we must be able to quantify the amount of contaminated water and, ultimately, the consequences of such contamination (*e.g.* in terms of human health impacts or ecological effects).

Nome of the above aspects can be determined with the necessary accuracy. But after some discussion with experts in the area (Forjaz, 1996), it was felt that even in the case of well rupture, the contamination of aquifers should be restricted to some limited time (perhaps less than one year). This because the most natural would be the progressive obstruction (scaling) of the geological communication channel(s) opened during the seismic event.

***Monetary valuation of the impact***

In view of the conclusions drawn in the previous section, this impact, although not negligible, is temporary. For a situation where the water requirements of a population of about 17,851 people (the total population of Ribeira Grande municipality) were not fulfilled, for the year 1990, this would correspond to a demand of about 1,045,833 $m^3$ of water (UNINOVA, 1991).

For this period (a one year period during which water resources stay improper for consumption), the most practical solution would be the transport of water from other regions into the island. Considering a transportation cost of about 1.37 ECU/$m^3$ of water, then an estimate of the resulting damage could be done.

In order to obtain a more precise evaluation of this impact, we took the values of the production at the seven water-springs that supply drinking water to the municipality. Those values are shown in Table 5.41.

**Table 5.41** Production of the seven water-springs that supply drinking water to the municipality of Ribeira Grande, Azores

| Water-spring designation | Production (l/s) |
|---|---|
| Ribeirinha/Monte Gordo | 14.4 |
| Matriz/Água Férrea | 10.0 |
| Conceição/Chá Canto | 20.0 |
| Ribeira Seca/Fajã Traves and M. Miguel | 17.0 |
| Sta. Bárbara/13 water-springs | 7.0 |
| Rabo de Peixe/Bandeirinha | 17.0 |
| Pico da Pedra/Fajã do Boi | 7.0 |
| Average | 13.2 |
| Total | 92.4 |

Only 36% of the water flow produced in the water-springs is consumed by the population of Ribeira Grande municipality (UNINOVA, 1991). One may then consider a more real situation of impact in which the water from a limited number of water-springs is contaminated by the geothermal fluid.

We can then introduce two probability factors *f1* and *f2*, accounting for the probability that an earthquake of magnitude over VII occurs during the power plant lifetime, causing multiple ruptures in the well (*f1*), and for the probability that such ruptures will affect the underground sources of water (*f2*). The probability product

$$f = f1 * f2$$

becomes the probability that an earthquake with a magnitude over VII be in the origin of water contamination problems during the period of one year (assumption).

Probability *f1* represents a cumulative probability, which means that if the annual probability of a seismic event VIII is 0.01, for instance, the occurrence of such a seismic event in the first year of a period of 100 years reduces the probability to zero in the remaining 99 years. In the same way, if a seismic event of magnitude over VII is noticed during 99 years, then the probability of occurrence of such an earthquake in the 100th year becomes 1.

Total damages ($T_d$) per year can therefore be calculated by the following expression:

$$T_d = \left( \sum_{i=1}^{N} (f * D)/(1+Dr)^i \right) / N$$

where $N$ is the number of years of the analysis (20 years), $D$ is the damage cost and $Dr$ the discount rate. Considering three scenarios of contamination of water-springs (see Table 5.42) and the respective probabilities (described above), an estimate of the impact can be obtain for three discount rates.

**Table 5.42** Scenarios under analysis

| Parameters | Scenarios | | |
|---|---|---|---|
| | A | B | C |
| Number of water-spring contaminated | 1 | 2 | 3 |
| Mean flow of each water-spring (l/s) | 13.2 | 13.2 | 13.2 |
| Contaminated water ($m^3$) | 416 275 | 832 550 | 1 248 826 |
| Drinking water ($m^3$) | 149 859 | 299 718 | 449 577 |
| Damage cost (ECU) | 205 253 | 410 506 | 615 759 |

**Table 5.43** Estimate of the impact damage costs for three different scenarios

| Parameters | Scenarios | | |
|---|---|---|---|
| | A | B | C |
| *N* | 20 | 20 | 20 |
| *D* | 149 859 | 299 718 | 449 577 |
| *Dr* | 0.1 | 0.03 | 0 |
| *f1* | 0.001 | 0.01 | 0.01 |
| *f2* | 0.1 | 0.3 | 0.5 |
| *f* | 0.0001 | 0.003 | 0.005 |
| Impact estimate | Low | Mid | High |
| Td (ECU/yr) | 8.74 | 916 | 3 079 |
| Damage (mECU/kWh) | 1.08E-04 | 1.14E-02 | 3.82E-02 |

Table 5.43summarises the scenarios considered for the assessment of this impact, as well as the final monetary estimates in mECU/kWh.

### 5.4.5 Impacts on water quality during heat mining

***Introduction***

During the drilling of the wells and the testing of the productive ones, underground waters that are normally used for farming, breeding and for other domestic or agricultural purposes can be affected and become improper for use. The constituents that may be introduced in ground and surface waters are the same that normally one can found in geothermal fluids. The physical and chemical characteristics of the geothermal fluid in our case study are shown in Table 5.35.

***Physical and monetary quantification of the impact***

This impact is temporary, thought to be limited to one year (assumption) and will be assessed using the methodology presented in the last section. The only difference introduced relates the probabilities considered. In the present case we assume that the probability factor *f1* has a value of 1 (the drilling activities, which represent the burden's source, are not probabilistic variables). On the other hand, the probability factor *f2* will be much higher (it is not dependent

**Table 5.44** Estimate of the impact damage costs during heat mining for three different scenarios (see text)

| Parameters | Scenarios | | |
|---|---|---|---|
| | A | B | C |
| *i* | 0/10 | 0/10 | 0/10 |
| *D* | 149 859 | 299 718 | 449 577 |
| *Dr* | 0.1 | 0.03 | 0 |
| *fl* | 1 | 1 | 1 |
| *f2* | 0.6 | 0.7 | 0.8 |
| *f* | 0.6 | 0.7 | 0.8 |
| Impact estimate | Low | Mid | High |
| Td (ECU/yr) | 9 147 | 25 490 | 49 261 |
| Damage (mECU/kWh) | 1.13E-01 | 3.16E-01 | 6.11E-01 |

purely on a random process, although some random component may be present). Thus, we assume that *f2* will approach 1.

Since this impact is rather expectable and burden's source will occurs with probability 1 during the years of drilling (year 0 and year 10), the expression to calculate the monetary values of the impact becomes:

$$T_d = [(f * D)/(1+Dr)^{i=0} + (f * D)/(1+Dr)^{i=10}] / 20$$

where i represents the years of drilling during the lifetime of the power plant (20 years).

The estimate of the impact damage costs during heat mining for three different scenarios (A, B and C) is presented in Table 5.44.

### 5.4.6 Impacts from geothermal fluid release

***Introduction***

Impacts of heat mining and waste disposal on surface (fresh and coastal waters) and groundwater quality can be significant. Contaminants such as boron, mercury and arsenic are often present in geothermal waters. When this water is discharged into water courses, the contaminated downstream waters can become polluted and thus be harmful to farming, fisheries, and other water supply purposes.

Thermal pollution can result from the direct disposal of insufficiently cooled brine. Despite in some cases an increase of water temperature can be beneficial, for example increasing biological production, on the other hand, increases in water temperature can have negative impacts. Many important commercial uses can be lost and some perturbations in ecosystem can occur. Higher temperatures can also increase the rate of oxygen depletion where oxygen-

demanding wastes are present, decrease oxygen solubility in water (Henry's Law) and settling velocity of particulates increases, because water viscosity and density decreases.

Dissolved oxygen depletion causes the decrease of biological respiration and can cause disturbances to fish populations, also because the demand of the organisms rises. In general all aerobic aquatic species are affected by decrease of dissolved oxygen. These disturbances to aquatic organisms are dependent upon the aquatic species and sensitive to the depletion on dissolved levels.

In general geothermal fluids present, in larger or in smaller quantity, some typical chemical constituents: chlorides, sodium, calcium, magnesium, potassium, aluminium, iron, bromides, manganese, strontium, boron, lithium, caesium, fluorides, rubidium, iodine, copper, arsenic, mercury, sulphates, silica, ammonium, carbon dioxide and rare gases; the radioactive elements are, in general, no much present either as number or as quantity: only radon is the most significant.

During mining, prospecting of geothermal fluids, and latter, during plant operation, waters which are normally used for farming, breeding, and domestic uses can be affected and become improper for consumption. The constituents that may be introduced in ground and surface waters are the same that normally one can found in geothermal fluids. The physical and chemical characteristics of the geothermal fluid in our case study are shown in Table 5.35.

Contamination of surface waterways and groundwaters must be avoided. Potential pollution can be prevented by reinjecting the separated brine back to the deep-lying geothermal reservoir. Another examples of mitigation measures for the brine destination and/or treatment are: (i) storing the brine in dry seasons with subsequent release to surface waters in the rain season; (ii) sedimentation processes; (iii) chemical precipitation; (iv) filtering; (v) inverse osmosis; (vi) electrolysis; (vii) ionic exchange.

***Contaminants of main concern (heavy metals)***

The following section presents a summary of some characteristics contaminants with particular concern in aquatic ecosystems, originated by direct brine discharge in aquatic ecosystems.

Natural waters contain a complex mixture of stable isotopes of trace metals, and many metals are essential for living organisms. Clark (1992), refer that metals of biological concern can be divided into three groups: (i) light metals (sodium, potassium, calcium and so on) normally transported as mobile cations in aqueous solutions; (ii) transition metals (e.g. iron, copper, cobalt and manganese) which are essential in low concentrations but may be toxic in high levels; (iii) metalloids (such as mercury, lead, tin, selenium and arsenic) which are generally not required for metabolic activity and are toxic to the cell at quite low concentrations.

According to the same author, there is considerable variation in the extent to which plant and animals can regulate the concentration of metals in the body: plants and bivalve molluscs are poor regulators of heavy metals; decapod crustaceans and fish are generally able to regulate

essential metals such as zinc and copper, but non-essentials such as mercury and cadmium are less well regulated.

The various factors influencing the toxicity of heavy metals in solution are compiled by Bryan (1976) *fide* Förstner & Wittmann (1981) (see Figure 5.19 hereafter).

The bioaccumulation of these contaminants depends on their biochemical properties and on the individual accumulation strategies of organisms for each element or isotope. Trace metals are taken up by aquatic organisms by more than one route, but the most important routes of uptake are those from solution and from food (Phillips and Rainbow, 1993). The solubility of metal compounds in water and in lipids is of great toxicological importance because it is one of the major factors influencing the biological availability and absorption of metals (Velimir Vouk, 1979).

Some heavy metals present in the brine produced by the geothermal power plant are of great concern to aquatic ecosystems. Based in works done by Förstner & Wittmann (1981), GESAMP (1990) and Clark (1992), some heavy metals of concern are discussed below.

*Mercury*
It is considered a non-essential but highly toxic element for living organisms. Even at low concentrations, mercury and its compounds present potential hazards due to enrichment in the food chain. Organic forms of mercury are more toxic than inorganic salts. Among metal contaminants, mercury is a special case because of the conversion of inorganic forms of mercury to persistent transfers to the top predators, with accumulation in fatty tissues over their longer life span.

*Cadmium*
This metal is not an essential element for any organism although, for unknown reasons, it enhances phytoplankton photosynthesis and growth at concentrations up to 100 ppm. Cadmium and its compounds, along with mercury and some other dangerous metals, are included in the "black list" of materials, which by international agreement may not be discharged into the sea. There is disputed evidence of accumulation of cadmium in marine waters through the food chain, although it is evidently taken by phytoplankton.

*Arsenic*
Although widespread in plant and animal tissues, arsenic has become synonymous with contaminant in public mind. In spite of its toxicity, it has been employed for its medical virtues in the form of organic arsenicals and in partial prevention of selenosis. Long term ingestion of arsenic-contaminated drinking water from wells has produced many health problems.

*Iron*
It is not usually a significant contaminant of the sea waters, but it has come to prominence over sea dumping of red mud from the extraction of alumina from bauxite and acid iron waste from the production of titanium dioxide.

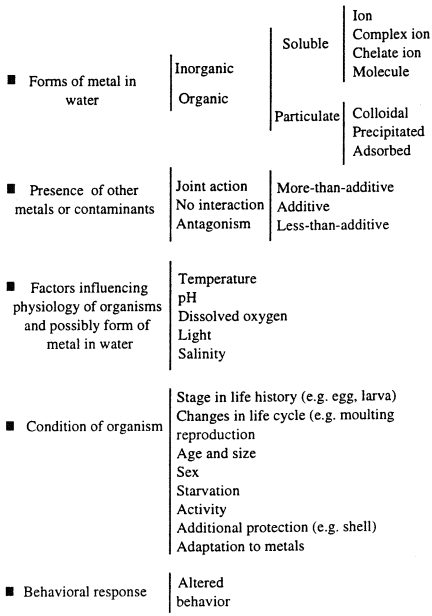



**Figure 5.19** Factors affecting the toxicity of heavy metals in solution.

The impact-pathway of brine discharges effects in the aquatic environment is illustrated in Figure 5.20.

***Quantification of the emission***

According to brine composition presented by UNINOVA (1991), the expected concentrations in water stream near the power plant are very high for many contaminants (Table 5.45).

With the lack available data is not possible to quantify the amounts of contaminants that arrives at coastal waters and ground waters.

***Physical assessment of the impact***

*Freshwater - Surface waters*
During prospecting stage will occur surface flows increases due to the release of the drilling geothermal fluid. The damp-proofing of the area associated to the power plant will produce an increment in runoff. The main water stream that crosses the area is the Ribeira do Teixeira, an affluent of Ribeira Grande, which receives the geothermal fluid rejected by the power plant.

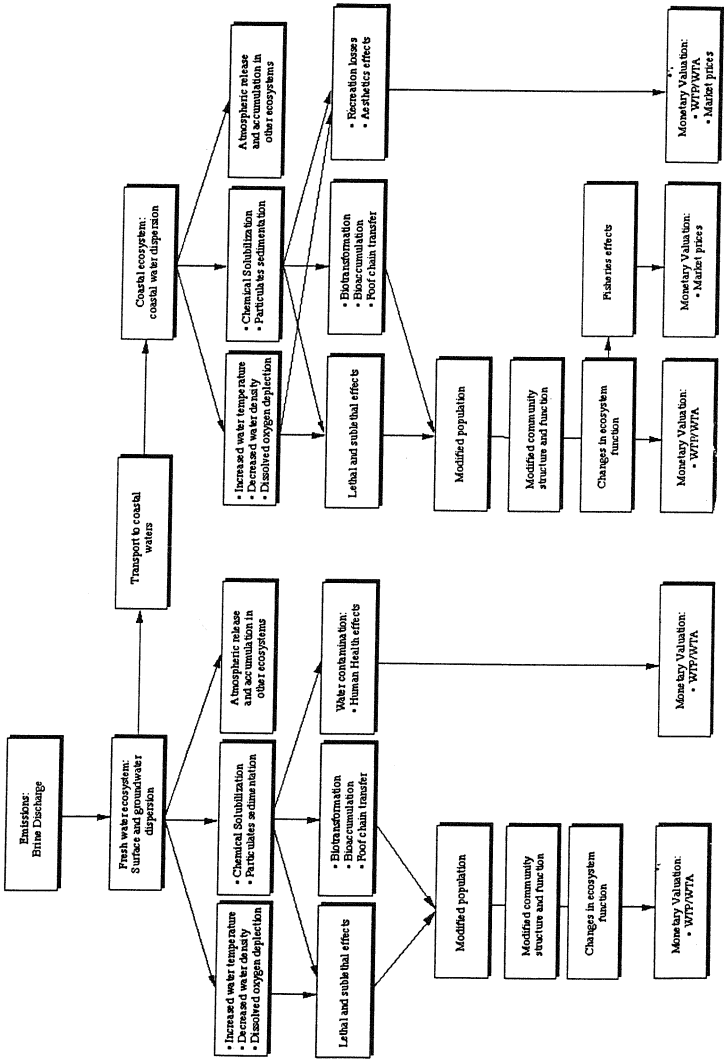



**Figure 5.20** Impact pathway of geothermal fluid release

*Groundwater - Base Aquifers*
No significant impacts are expected to affect the base aquifer during the drilling phase. On the contrary, there will be important impacts in water quality and quantity of the base aquifer during the operating phase. With an extraction flow of 490 $m^3$/hour the depression originated around the extraction point will originate a rise level of salt and brackish water in the reservoir.

*Coastal waters*
There is no monitoring program of the coastal waters that receive the released geothermal fluid transported by the water streams. However, it can be assumed that significant amounts of chemical compounds can reach the coastal zone and originate damages in coastal ecosystem. Despite the potential strong hydrodynamics and consequent high level of dispersion, contaminants, even in low concentrations, can accumulate at high ecosystem levels by bioaccumulation and food chain transfer and produce changes in the biological systems. Also mechanisms of sedimentation of particulate matter can store contaminants in bottom sediments that later supply the water column and biota.

To evaluate the impact in coastal ecosystems, it would be necessary to determine water contaminant dispersion and consequent concentrations obtained, with the aid of mathematical models. On the other hand to estimate the ecological consequences of these concentrations two approaches could be used: (i) simulation of ecosystems processes with ecological mathematical model; (ii) ecological risk assessment based on methodologies like presented by Suter II (1992), USEPA (1992), Suter II (1995), Suter II *et al.* (1995).

Ecological risk assessment is the application of risk assessment to risks to non-human organisms, populations and ecosystems. In opposition to the restrict protection of human, ecological risk presents: (i) some routes of exposure are unique to ecological receptors, (ii) some ecological receptors are likely to be more sensitive; (iii) some mechanisms of action are not relevant to humans; (iv) exposure is more intense; (v) nonhuman organisms are more subject to secondary effects.

The ecological risk assessment consists of four internal interrelated tasks (LaGrega *et al.*, 1994): (a) Characterisation of the baseline ecology and identification of potential receptor populations;(b) Ecological toxicity assessment; (c) Ecological exposure assessment; (d) Risk characterisation.

Besides the absence of original data for the characterisation of the coastal ecosystem and contaminant amounts that arrive at the water system, there is a general deficit of adequate simulation models for ecological impact analysis. Although some models have been already developed and tested for coastal ecosystems around the world, some critical problems remain when we try to transpose these models to different ecosystems, mainly due to the singularities and complexity that characterise coastal processes.

*Monetary valuation*

For coastal systems, four types of damages can be present:
(i) Ecological - a monetary valuation can be done by applying of the contingent valuation method (European Commission, 1995a): *willingness to pay* (WTP) or the *willingness to accept* (WTA);
(ii) Fisheries (commercial and recreational) - a monetary valuation can be done by the use of market prices;
(iii) Water-based recreation - a monetary valuation can be done by the application of market prices;
(iv) Aesthetics monetary - a monetary valuation can be performed using a contingent valuation method.

For freshwater systems two types of damages are present:
(i) Ecological - a monetary valuation can be done by the use of a contingent valuation method;
(ii) Human health effects - a monetary valuation can be performed using a contingent valuation method.

Giving the lack of original data and adequate damage functions, it is not possible to estimate with an acceptable confidence level the costs related to this impact. There are many difficulties in using existing valuation studies to estimate impact on water amenity. One of the reasons for that are the pathways by which water is affected are complex and not always sufficiently known, especially in terms that makes them amenable to valuation.

### 5.4.7 Impacts from induced seismicity and/or subsidence on constructed buildings and facilities

*Introduction*

Ground subsidence can be caused by withdrawing large amounts of subterranean water from a hydrothermal field. Ground subsidence can damage surface and sub-surface structures such as pipelines, water and wastewater drainage systems, and general buildings. These impacts can be quantified directly by evaluating the property damages involved.

Reinjection of the geothermal fluid as well as using different technologies for heat exploitation (*e.g.* using down-hole heat exchangers instead of extracting the natural thermal fluids) are thought to be good solutions to avoid ground subsidence near the producing wells. However, the option of reinjecting the fluid has been thought to induce seismicity in critically stressed areas. Small-scale seismic events can also damage surface and subsurface structures. Such events can be either associated with subsidence phenomena or act alone as a destructive agent.

Despite the positive aspects relating fluid reinjection, which include the overcoming of some potential water pollution and waste disposal problems, there have been pointed out some further undesirable aspects such as (1) the possible interference with the useful heat output from

a producing well; (2) the excessive power requirements to pump down the fluid, and (3) the possibility of decreasing underground permeability by chemical deposition.

Induced seismicity and subsidence may to a certain extent be considered *alternative* impacts depending on whether or not geothermal fluid reinjection takes place. In general, fluid reinjection substantially provides for the reduction of the possibility of ground subsidence around the wells area. On the other hand, induced seismicity (the record of microearthquakes) has been sometimes associated to fluid reinjection .

Research done in a geothermal field in Iceland, and reported by Kristmanddsdóttir and Ármannsson (1995), showed that the injection pressure into the studied field was far below that needed to induce seismicity. Drowdown decreased pore pressure and thus increased rock strength and reduced microseismic activity. The same authors reported also conclusions of ground levelling and gravity measurements. Subsidence rates measured in different sites were found to be in the range of 5-14 mm/year.

***Quantification of the burden***

At the moment, no reinjection takes place in the geothermal power plant under study, meaning that there might be a major concern regarding subsidence impacts in the neighbourhood of the wells. Yet, no information was up to now reported on subsidence cases in the vicinity of the well.

The area potentially affected by subsidence should in any case exceed a radius of 500-1000 meters from the existing CL1 well. This means an area in risk between 78 and 314 hectare.

Seismicity will depend mostly on the geologic structure crossed by the well and on the rate at which the geothermal system is recharged. If the mass output from the well is higher than the recharge rate to the system, then the system is unbalanced and therefore subsidence is expected. The major problem here consists of how to define the boundary of the geothermal system: in fact, one may have a system which is mass-balanced for a sufficiently large spatial domain, whereas it is clearly not at a local scale.

Despite the above scale problem, one may say that subsidence will always be directly related to the volumetric flow extracted from the geothermal system. This flow was estimated by considering the geothermal fluid flow rate that is rejected by the plant after power generation. Such flow rate is about 0.14 $m^3/s$ (UNINOVA, 1991), which is equivalent to 4.4 $Mm^3$ per year. This volume of rejected fluid can be assumed as roughly equivalent to that which is extracted from the geothermal system, once considered its pressure and temperature conditions.

***Physical and monetary quantification of the impacts***

Subsidence rates are assumed to be in the range 0 - 10 mm/year, thus we admit that for a optimistic scenario there will be no impact at all due to land subsidence.

Considering the well's lifetime, for the worst scenario we would therefore obtain in the vicinity of the well, and for those more sensitive areas, maximum subsidence levels of about 10 cm.

Located in a wild area (actually, a natural Park), at an altitude of about 530 meters, the power plant is relatively far from dwellings and transport facilities such as roads or water and wastewater drainage systems. Thus, even if the burden was significant, the final physical impact would result negligible, because of the absence of valuable endpoints in the neighbourhood of the power plant.

### 5.4.8 Impacts of air pollutants on human health, local amenity, and terrestrial ecosystems

***Introduction***

The atmospheric emissions of concern from geothermal fuel cycle are those of $CO_2$, which was already addressed in section 5.4.3, hydrogen sulphide ($H_2S$), mercury and arsenic compounds and traces of radioactive elements such as radon (produced by the decay of radium) and lead (Pb -210). Yet there have been reported emissions of ammonia, boric acid, hydrogen, methane, fluorides and traces of NOx and particulates.

Carbon dioxide may represent a local hazard in that it may pocket in low lying areas (Dench, 1988). This seems to be an impact that only occurs in very particular sites, with climatic and topographic characteristics far different from those of our case study.

Geothermal fluids almost invariably contain some amount of hydrogen sulphide, which has been sometimes reported as an air pollutant of major concern (Bloomer, 1995; Kristmanddsdóttir and Ármannsson, 1995).

In low concentrations, $H_2S$ has an unpleasant smell. However, when strongly concentrated it paralyses the olfactory nerves and thus becomes odourless and dangerous. Being heavier than air, it tends to collect in low-lying pockets. Human health fatalities are known to have occurred in the vicinity of fumaroles, but no such cases have so far been associated with geothermal exploration or power production. In fact, it has been suggested that in general $H_2S$ very rarely creates a health risk (Fridleifsson and Freeston, 1995). Figure 5.21 compares sulphur emissions from geothermal and fossil fuel-fired power stations.

***Quantification of the emissions***

The total emissions of non-condensable gases of concern regarding this impact in the geothermal plant under study are given in Table 5.46.

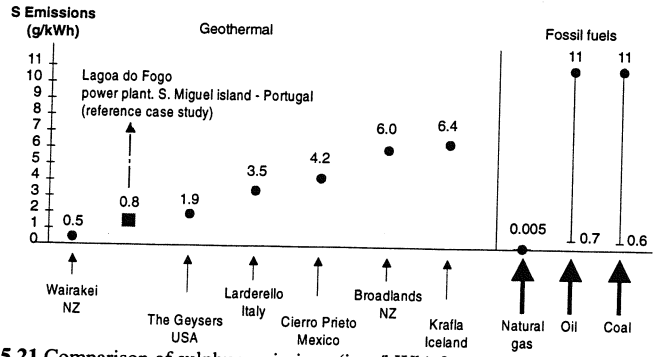



**Figure 5.21** Comparison of sulphur emissions (in g/kWh) from geothermal and fossil-fired power plants (Sources: Fridleifsson and Freeston, 1994; European Commission, 1995b-c).

**Table 5.46** Atmospheric emissions of non-condensable gases at Lagoa do Fogo geothermal power plant.

| Pollutant | Emissions (g/kWh) |
|---|---|
| $H_2S$ | 0.81† |
| Other gases | 1.62 |
| Residual steam | 397 |
| n-pentane (loss of motive fluid) | 0.002 |

†: This corresponding to an annual emission of about 65.2 ton $H_2S$ /year.

***Physical quantification of the impact***

Hydrogen sulphide - with the notorious 'bad eggs' odour - is a highly toxic and flammable dense gas. There are few industrial uses for $H_2S$ but it is produced in large amounts as a by-product of a number of both natural and industrial processes. In geothermal developments, the source of $H_2S$ is the sulphur-containing rock.

In urban areas, levels of $H_2S$ are normally much less than 5.4 ppbv. If they exceed this level for any length of time then substantial complaints are likely to be made by the public because people's sense of smell with respect to this gas is so acute. The threshold of human odour detection occurs between 0.15 and 1.5 ppbv, whilst the rotten egg smell is clearly apparent at about three times this level.

Hydrogen sulphide is slightly phytotoxic. Only at levels above 100 ppbv have necrotic lesions and tip-burn been observed on a variety of plants. Indeed it may be one of the few other instances apart from carbon monoxide where plants are less sensitive to an atmospheric pollutant than man himself. Beneficial effects upon growth have been reported with low concentrations (30 ppbv) but, of more interest, is the fact that living (as well as rotting) vegetation may be a significant source of $H_2S$ (Wellburn, 1988).

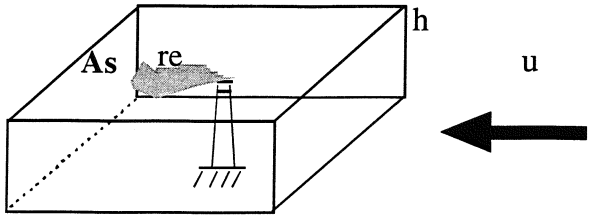



**Figure 5.22** Illustration of the box model used to predict $H_2S$ concentrations in the vicinity of the power plant (*h* represents the height of the mixing layer).

Thus, and as far as $H_2S$ is concerned, the impacts of atmospheric emissions from geothermal developments are likely to occur upon human health or, at least, local amenity (because of possible malodorous emissions). Impacts upon local vegetation are considered negligible because of the weak sensitivity of plants to the gas. But this conclusion was further supported by a rough evaluation of the concentrations expected in the vicinity of the power plant during its normal operation.

*Evaluation of the $H_2S$ concentrations in the vicinity of the power plant*

A rough evaluation of the pollutant concentrations in the vicinity of a point source can be done through a mass-balance evaluation to an hypothetical control volume (box) *V*. Assuming instantaneous mixing within such a volume[1], the evolution of the mass *M*, of a conservative pollutant is given by:

$$dM/dt = r_e - ((A_s u^*)/V) * M \quad (1)$$

where $r_e$ is the mass input rate of the pollutant into the control volume (g/s), $A_s$ is the area perpendicular to the wind direction ($m^2$), $u^* = u + u'$ is the advective component of dispersion, which lumps wind velocity $u$ and an additional velocity component u' accounting for turbulent diffusion.

[1] This assumption is the most restrictive for the application of a box model. At a distance of, say, 5km from the souce, the value of the horizontal dispersion coefficient is, for the Pasquill-Guilford stability class D (neutral), one order of magnitude lower than the grid-cell size. The meaning of this is that might be places, in certain atmospheric conditions, with higher pollutant concentrations, whereas in other places (e.g. upwind from the source) such concentrations will be much lower. For a sufficiently great period of integration, one may say that such differences will be smoothed and thus the gradient of concentrations within the control volume will become very small.

**Table 5.47** Model parametrization

| | Dispersion scenarios | | |
|---|---|---|---|
| | A | B | C |
| h (m) | 300 | 600 | 1000 |
| u* (m/s) | 1.5 | 2.5 | 4 |
| re (g/s) | 2.43 | 2.43 | 2.43 |

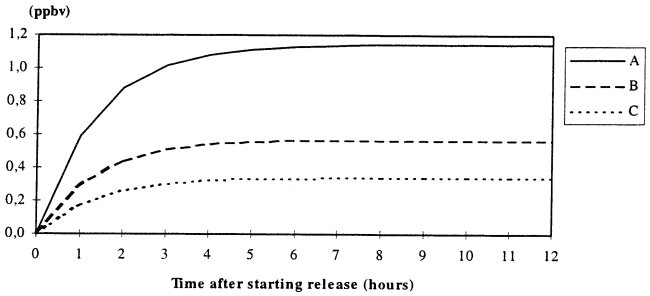



**Figure 5.23** Predicted $H_2S$ concentrations in the control volume[2]

Equation (1) can be discretized using an explicit finite-difference[3] scheme thus yielding:

$$M^{t+dt} = M^{t} + (r_e - M^{t} (A_s u^{*t})/V)\ dt$$

The parametrization of the model was done for three scenarios of dispersion using a bidimensional grid spacing dx=dy=5 000 m. The scenarios were established in order to obtain figures on the probable most adverse, medium, and optimal dispersion conditions. Table 5.47 summarises the three scenarios considered.

*Results*

The results obtained using an integration time step of 200 s for the three different scenarios are depicted in the next figure. For the three scenarios equilibrium concentrations are achieved after about 6 hours of simulation, and are of 1.14, 0.57 and 0.34 ppbv for the scenarios A, B and C, respectively.

*Assessment of human health impacts*

Table 5.48 summarises the effects of $H_2S$ on human health for different exposure ranges. Some of these effects have already been quantified from the monetary point of view (*e.g.* eye irritation) in previous ExternE studies. However the expected concentrations in our hypothetical box volume are lower than 1.2 ppbv, meaning that for the worst scenarios there will be

[2] (1 ppbv $H_2S$ = 1.417 μg $m^{-3}$ at 20°C).

[3] This numeric method is stable whenever the Courant number C = u dt/dx is lower than 1. For C=1 we obtain the exact solution (dx represents the grid spacing).

**Table 5.48** Summary of the effects of hydrogen sulphide on humans (adapted from Wellburn, 1988).

| Exposure range | Effects and symptoms |
|---|---|
| 0 - 0.2 ppbv | No discomfort |
| 0.2 - 1.2 ppbv | Odour detected |
| 1.2 - 5.4 ppbv | Consciousness of 'bad egg' smell |
| 5.4 ppbv - 10 ppmv | Eye irritation threshold |
| 10 - 50 ppmv | Severe eye irritation and impairment |
| 50 - 100 ppmv | Characteristic eye damage called 'gas eye' |
| 100 - 320 ppmv | Loss of sense of smell, nausea, and increased lung irritation |
| 320 - 530 ppmv | Lung damage and water accumulation |
| 530 - 1000 ppmv | Shortage of breath, stimulation of respiratory centre, convulsions and chance of respiratory arrest |
| 1000 ppmv and above | Immediate collapse and respiratory failure immediately before death |

some odour detection, although probably not perceived as annoying: the concentration thresholds for consciousness of 'bad egg' smell and for physiologic symptoms are not reached according to the model results.

***Monetary valuation of the impact***

Given on the one hand the expected concentrations of $H_2S$ in the vicinity of the power plant, and, more important, the absence of potential receptors on the other, one may say that this impact is likely to be negligible, both from physical and monetary points of view.

Furthermore, according to the Odour Impact Model concept, as presented by Nagy (1991), the 50% detection level[4] for $H_2S$ is of 5.5 $\mu g\ m^{-3}$ (3.9 ppbv), which is a value 3.4 times higher than the value predicted in our modelling exercise for the worst scenario of dispersion.

### 5.4.9 Occupational health impacts

***Introduction***

According to European Commission (1995c) occupational health impacts area at least partially internalised through insurance, or wage premia for jobs with greater risks than the average. Occupational diseases generally occur as a more or less delayed response to a continuous exposure of an external burden (e.g. airborne pollutants, noise, vibration)

For the geothermal fuel cycle, occupational health impacts occur during drilling operations for the construction of geothermal wells. And as for other fuel cycle occupational accidents could occur during the power plant construction, operation and dismantling.

[4] 50% of a sample of human population (odour panel) detects the odour.

**Table 5.49** Occupational impacts estimated during drilling operations

| Accident category | Drilling operations (accidents per year) |
|---|---|
| Fatal accidents | 0.0014 |
| Serious accidents | 0.0332 |
| Minor accidents | 0.2300 |
| Sub-total | 0.2646 |

The prevision of the plant duration time is about 20 years, but this value is a little uncertain and can be grater. The dismantling of the geothermal plant will probably be determined by the low production of the wells and not because of the deterioration of the equipment. The technical lifetime of the each well is estimated to be about 10 years.

If the wells production becomes low, with the risk of reducing electricity generation, there is the possibility of drilling new wells near the actual area or moving the plant to another site where geothermal fluid is available.

***Methodological approach***

Giving the lack of information on accidents and injuries from this activity, we have to rely on existing information from other fuel cycles with similar activities, namely oil and gas cycles. Information taken from European Commission (1995c), gives different accidents rates for each accident category: (i) Fatal accidents: 0.0036 (per well); 0.00016 (per TWh); (ii) Serious accidents: 0.083 (per well); 0.0038 (per TWh); (iii) Minor accidents: 0.575 (per well); 0.026 (per TWh).

For the other stages of the fuel cycle, power plant construction, operation and dismantling is not possible to estimate the impacts due the lack of data. However is presented some examples for natural gas in UK that could present a possible related scenario.

***Physical quantification of the impact***

Assuming the rates for each accident category during drilling operations and the construction of the 4 wells each 10 years of activity, a number of occupational impacts has been estimated (see table hereafter).

***Monetary quantification of the impacts***

The valuation data for the assessment of accidents are presented in the ExternE report, on methodological issues (European Commission, 1998). The following values in ECU(1995) have been used: (i) fatalities: $3.14 \times 10^6$ ECU; (ii) serious damage: 95,050 ECU ; (iii) minor injury: 6,970 ECU.

**Table 5.50** Occupational impacts from geothermal fuel cycle (mECU/kWh)

| | Fatal accident | Serious accident | Minor accident | Total |
|---|---|---|---|---|
| Drilling | 5.61E-05 | 3.91E-05 | 1.99E-05 | 1.15E-04 |
| Construction | 7.85E-03 | 7.47E-03 | 2.72E-03 | 1.80E-02 |
| Operation | 2.05E-03 | 2.31E-03 | 1.83E-03 | 6.19E-03 |
| Dismantling | 4.11E-04 | 3.73E-04 | 1.28E-04 | 9.12E-04 |
| Total | 1.04E-02 | 1.02E-02y | 4.69E-03 | 2.53E-02 |

In order to present some estimates of the impacts from the other stages of the fuel cycle, data on occupational accidents from other fuel cycle studies (European Commission, 1995c) have been used. Giving the relative homogeneity of the results presented, we can consider these values as a rough estimate for the geothermal fuel cycle. Table 5.50 summarises the results on occupational accidents from the geothermal fuel cycle.

Total estimated impacts from occupational accidents are relatively low and reach 0.025 mECU/kWh.

### 5.4.10 Review of other impacts

*Aesthetic impacts*

As geothermal energy sources are often found in non-industrial areas, this impact can be very important in the global balance of externalities. Access roads, drilling rigs, pipes, cooling towers and other power plant facilities can be the major elements of intrusion in the landscape.

In general, previous to the establishment of a geothermal development there is a phase of heat field prospecting during which a great amount of aesthetic impacts can occur. These aesthetic impacts, together with the impacts of the power plant facility construction itself, are temporary and thus their overall magnitude must be weighted in time (considering both the lifetime of a given geothermal hole or the lifetime of the geothermal power plant). The major outcomes of these impacts are associated to interference with the sensorial perceptions of the people who pass in the construction or prospecting areas, as a result of the spatial and functional disorganisation of the affected space. The main potential burdens during this phase are:

- Introduction of elements not conform with the traditional environment (e.g. heavy machinery and building materials);
- Decreased visibility in the construction sites because of the increment in the concentration of dusts in the atmosphere, which deposit in the surrounding areas changing their visual appearance.
- Changes in the visual character of the sites, because of the alterations in land uses and functions, and with the removal and change of characteristic landscape elements.
- Introduction of artificial elements of large dimensions and of difficult visual integration.

During operation some of the impacts identified during construction are eliminated. However, the impacts related to the presence of artificial elements in the landscape, as well as the impacts from changes in the terrain morphology, become definitive.

*Valuing aesthetic impacts*

Aesthetic impact valuation is inevitably a subjective task. However, for areas relatively undisturbed by man's activities, it seems consensual that the aesthetic achievements of nature are nearly always more desirable than any human (subjective) interference. Thus it would be possible to evaluate this impact by asking to a representative panel of affected people their preferences for things they desire, and their willingness to pay (WTP) for the non-construction of a given power plant. This would be the direct contingent valuation (CVM) approach.

Once established the construction of a given geothermal power plant, the CV method can still be applied, by asking the people what they are willing to receive by way of compensation to tolerate the presence of the project's elements (WTA). In this case the answers should not be so homogeneous as for the case of an *a priori* evaluation, in part because of the subjective aesthetic perception of the (given) proposal for development.

European Commission (1995d) reviews three studies relevant for the evaluation of visual amenity damages: (1) Hanley (1989), (2) Willis and Garrod (1991), and (3) Hanley and Ruffel (1993).

The first study is primarily a comparison of two valuation techniques (TCM and CVM) applied to the Queen Elizabeth Forest in Central Scotland. Parts of the Queen Elizabeth Forest lie in the Loch Lomond and Trossachs National Scenic Areas of Scotland which are the closest equivalents to National Parks.

The second study is the only one devoted entirely to the problem of valuing landscape amenity, and covered both residents and visitors in the Yorkshire Dales National Park. The area of the case study is an upland region, with a very high quality landscape and exploited mainly with recreational purposes by the large populations in the adjacent urban areas of Northern England.

The third study also relates to sites for forest recreation, but surveyed visitors to a larger number of Forestry Commission sites. The objective of the study was not only to find the WTP for forest recreation, but also the forest characteristics to which the WTP is sensitive. Rating of the quality of views was one of the parameters analysed.

Despite the importance of these studies for the quantitative assessment of aesthetic impacts, they are obviously site-specific and the transferability of the results obtained for a particular case study would be very difficult and doubtfully performed to evaluate the impact of another particular situation. The wild nature of the area surrounding Lagoa do Fogo geothermal power plant, which is actually a natural Park, suggests that a CV study should be made accounting for its particular features. Time and budget constraints forced us to left such evaluation out of

the scope of this study, hoping that future research will complement our efforts on the quantification of the externalities of geothermal fuel cycle.

***Noise***

Two types of noise associated with geothermal developments were identified:
(i) Temporary noise, generated during drilling and construction of the different constituting parts of the project (geothermal wells, power plant facilities, permanent and temporary roadway accesses). Includes also the noise due to transportation of materials and people to the construction/operation sites.
(ii) Permanent noise during plant operation, mainly due to steam escape and to the operation of other mechanic components of the installed system.

*Physical quantification of the impact*
Sound propagation physical models can be used to determine the sound levels expected at some distances from a source of sound, and in certain conditions. A review of these numerical models can be found in Mestre & Wooten (1980).

Noise has many effects upon humans, and these can be categorised in as many different ways as there are categorisers. One listing of noise effects would separate them into 5 categories: (i) damage to the hearing system, (ii) impairment of hearing, (iii) task performance disruptions, (iv) non-acoustic physiological effects, and (v) annoyance.

In general, the greater the intensity and the longer the duration of noise, the greater is the likelihood of the various sorts of effects. This relationship is clearest under very quiet or under very noisy conditions. The fact that several types of effects appear to be relatively monotonic, however, does not imply that the functional relationships relating magnitude of noise to magnitude of effect are the same. Indeed it is not even the case that the same parameters of the noise exposure are operative for different classes of effects (Singer & Baum, 1987).

Several studies and reviews have been made on human response to noise (e.g. Glass & Singer, 1972; Kryter, 1985). Yet, some theoretical models have been proposed (e.g. Singer & Baum, 1987). However, these models can only be applied in certain controlled conditions, and if a certain amount of information is available. Moreover, since people can and do respond to combinations of stimuli, is always very difficult to disentangle between purely noise effects and effects caused by combinations of noise and other stimuli.

Another characteristic of some sources of noise, such as transportation systems, is its intermittence. In this case the emission of noise is not continuous, and the overall effects must be integrated in order to attain a representative quantification.

Some approaches consist of aggregating noise effects into overall indexes. A first approximation to an overall index can be a general linear model in which standardised amounts of effect from each source is used as a predictor of total effects. Some approaches seem to be particularly suitable for the quantification of intermittent noise impacts, as well as for the determina-

tion of the external costs of noise. The noise nuisance index (NNI: ETSU, 1993), for instance, seems to be an adequate method for the quantification of rail and transportation traffic noise.

*Monetary valuation of noise impacts*
Noise impacts are not expected to attain for a significant percentage of the global impacts of geothermal fuel cycle. Yet, when considering Lagoa do Fogo case study, such impacts are thought to be negligible given the non existence of human settlements in the area surrounding the power plant and the heat prospecting areas. Possible impacts are those affecting local fauna that can be temporarily disturbed by the presence of artificial elements in the developing areas. However, no methodologies are available at the moment allowing the quantification of such impacts, and thus this parcel should remain unclosed for future developments of ExternE project.

### 5.4.11 Conclusions

This final section documents the achievements of the project undertaken under the ExternE accounting framework on "External cost of geothermal fuel cycle". The methodology adopted is based on the impact-pathway analysis or damage function approach where activities are linked to emissions and related impacts. In this project, a bottom-up approach is used to quantify external costs (and benefits) of fuel cycles, using the existing information and available methods.

The final results obtained for the geothermal fuel cycle are synthesised in Table 5.51. A first conclusion from our analysis on a geothermal plant located at the São Miguel island should emphasise the importance of global warming impacts in the global accounting of externalities (this conclusion is very site/resource dependant and should not be considered valid for other cases of energy generation from a geothermal source). As a matter of fact the relative weight of such impact was in our case of about 97-99%, using the restricted range. Even for the lowest estimate of global warming damage costs using a 95% confidence interval, the relative weight of global warming impacts would account for about 91% of the externalities of geothermal fuel cycle. This seems to be one of the most relevant conclusions of our study since it stresses the question of whether or not geothermal sources of energy should be integrated in the same category as renewables emitting marginal levels of GHG.

A great number of impacts from geothermal fuel cycle cannot be assessed using the current version of Ecosense since atmosphere is not the major pathway through which the pollutants are transported up to the receptor(s). Water quality impacts are rather important for this fuel cycle and, although some of them not quantified in this study (*e.g.* the impacts of geothermal fluid release on aquatic ecosystems), water quality impacts due to heat mining generate the second most important externality of our evaluation. The associated damage, valued in about 0.3 mECU per kWh, seems quite relevant in the global balance of externalities.

Under the classification of *other impacts* we have included noise and aesthetic impacts. Although not quantified it was felt that their relative weight in the global accounting of external costs would be rather small. The same happened with impacts from induced seismicity and

**Table 5.51** Summary of results for the geothermal fuel cycle

| Impact | Damage cost | |
|---|---|---|
| | mECU/kWh | ECU/Unit of pollutant |
| Global warming impacts due to greenhouse gas emissions: | 14.4-37* | 18-46 ECU/t of $CO_2$ equivalents emitted |
| Impacts on water quality due to seismic and volcanic activity: | 0.0114** | - |
| Impacts on water quality due to heat mining: | 0.316** | - |
| Environmental impacts from geothermal fluid release: | <0.001 | - |
| Impacts from induced seismicity or subsidence on buildings and infrastructures: | <0.001 | - |
| Impacts of $H_2S$ emissions on human health and terrestrial ecosystems: | <0.001 | <1.23 ECU per t of $H_2S$ |
| Occupational health impacts: | 0.012 - 0.034 | - |
| Other impacts: | <0.001 | - |
| Sub-Total: | 14.7 – 37.3 | - |
| TOTAL: | nq | - |

nq: not quantified
* Illustrative restricted range.
** Mid estimates.

subsidence on constructed buildings and infrastructures. One of the reasons for this conclusion was the absence of dwellings, public service facilities or other buildings of public interest in the neighbourhood of the power plant[5].

The impacts of hydrogen sulphide on human health were discussed and a box model was used to predict the concentrations of the compound in the vicinity of the power plant. The results showed that, even in the worst conditions of dispersion, the concentrations are lower than the limits of human detection. Thus this impact was considered negligible.

Occupational health impacts were quantified for the different stages of the fuel cycle using data presented in previous ExternE studies. The orders of magnitude of the results obtained are not far from the threshold of what should be considered a negligible impact. However we would like to stress that the estimates presented herein did not take into account the real situation and characteristics of geothermal fuel cycle, and thus they should be regarded as rough estimates.

Despite all the progress made on impact quantification, some impacts still remain non-quantified either from the physical or from the economic points of view. The reasons for that are multiple and include:

[5] Although this can be subject of criticism since the anthropocentric nature of the ExternE accounting methodology may suggest that those things that are outside the sphere of human knowledge have no economic value.

- lack of knowledge and models (process description tools) for some areas like ecology and medium-long term ecological effects;
- lack of data for the quantification of burdens, physical impacts or economic valuation of damages and benefits;
- lack of dose-response functions in some areas (*e.g.* ecological responses as valuable end-points);
- lack of monetary values or values directly transferable from one context to another.

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# A THE ECOSENSE MODEL

## A.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.

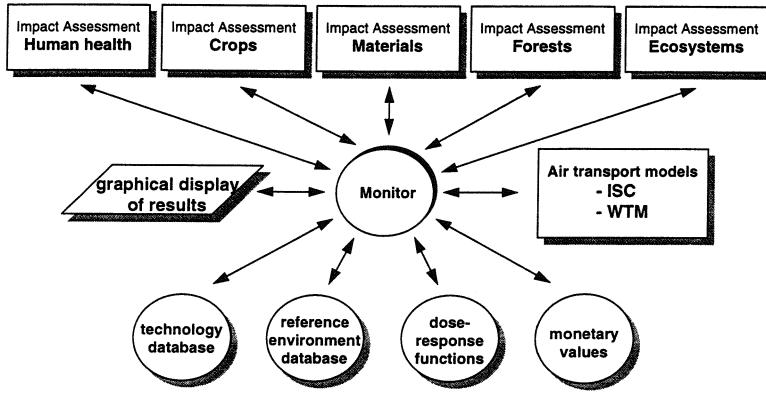



**Figure A.1** Structure of the EcoSense model

## A.2 Scope of the EcoSense model

EcoSense was developed to support the assessment of priority impacts resulting from the exposure to airborne pollutants, namely impacts on health, crops, building materials, forests, and ecosystems. Although global warming is certainly among the priority impacts related to air pollution, this impact category is not covered by EcoSense because of the very different mechanism and global nature of impact. Priority impacts like occupational or public accidents are not included either because the quantification of impacts is based on the evaluation of statistics rather than on modelling. Version 2.0 of EcoSense covers 13 pollutants, including the 'classical' pollutants $SO_2$, $NO_x$, particulates and CO, as well as some of the most important heavy metals and hydrocarbons, but does not include impacts from radioactive nuclides.

## A.3 The EcoSense Modules

Figure A.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.

### A.3.1 The EcoSense database

#### *A.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.

#### *A.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.

#### *A.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.

#### *A.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

### A.3.2 Air Quality Models

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

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

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

### A.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.

### A.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.

## A.4 The air quality models integrated in EcoSense

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

Close to the plant, i.e. at distances of some 10-50 km from the plant, chemical reactions in the atmosphere have little influence on the concentrations of primary pollutants, if NO and its oxidised counterpart $NO_2$ can be summarised as $NO_x$. Due to the large emission height on top of a tall stack, the near surface ambient concentrations of the pollutants at short distances from the

stack are heavily dependent on the vertical mixing of the lower atmosphere. Vertical mixing depends on the atmospheric stability and the existence and height of inversion layers (whether below or above the plume). For these reasons, the most economic way of assessing ambient air concentrations of primary pollutants on a local scale is a model which neglects chemical reactions but is detailed enough in the description of turbulent diffusion and vertical mixing.

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

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

where:

| | | |
|---|---|---|
| | $c(x,y,z)$ | concentration of pollutant at receptor location $(x,y,z)$ |
| | $Q$ | pollutant emission rate (mass per unit time) |
| | $u$ | mean wind speed at release height |
| | $\sigma_y$ | standard deviation of lateral concentration distribution at downwind distance $x$ |
| | $\sigma_z$ | 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 $\sigma_y$ and $\sigma_z$ 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.

### A.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 A.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).

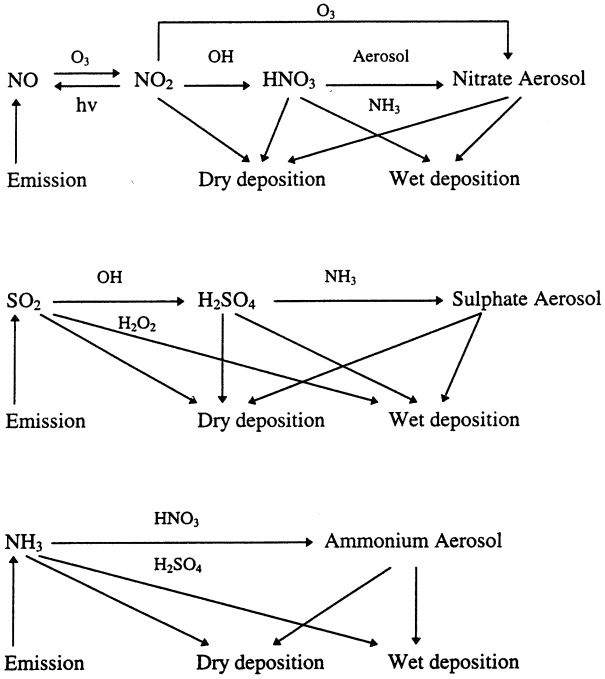



**Figure A.2** Chemical Scheme in WTM, adopted from Derwent et al. (1993)

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# B ANALYSIS OF HEALTH EFFECTS

## B.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.

## B.2 Non-Carcinogenic Effects of Air Pollutants

### B.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.

### B.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 B.1 and Table B.2. Further details on the uncertainty classification given in the final column of the table are given

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

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

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

[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 B.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-$\mu g/m^3$)] for morbidity, and [%change in annual mortality rate/($\mu g/m^3$)] for mortality.

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

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

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

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.

## B.3 Carcinogenic Effects of Radionuclide Emissions

### B.3.1 Introduction

A brief explanation of the terminology specific to the nuclear fuel cycle assessment is presented in Box B.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.

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.

**Box B.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.

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.

### B.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.

### B.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 B.1.

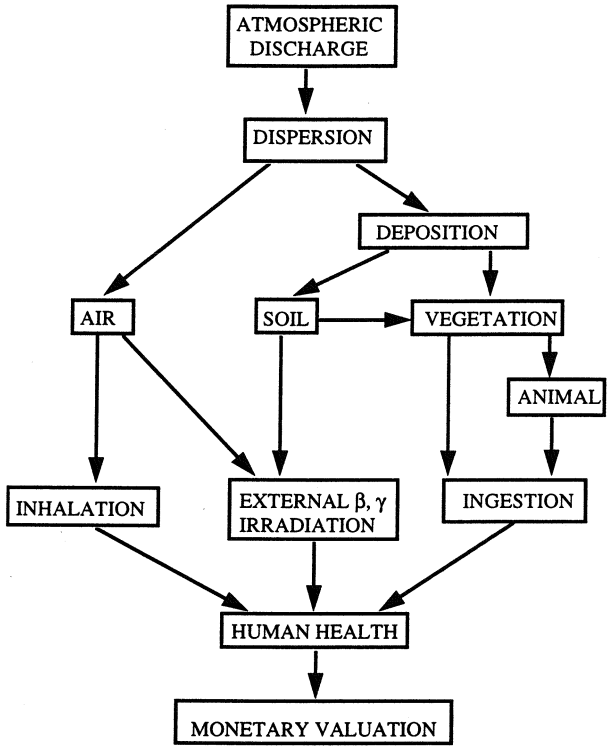



**Figure B.1** Impact pathway for an atmospheric release of radionuclides into the terrestrial environment

### *B.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).

### *B.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 B.3.

**Table B.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 |

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 B.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.

### *B.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 B.5.

**Table B.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 |

**Table B.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 |

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

The fraction of cancers that would be expected to be non-fatal (0.12 non-fatal cancers per manSv) are calculated based on the expected number of fatal cancers and the lethality

fractions reported for 9 categories of cancer in ICRP60. This is reflected in the aggregated non-fatal cancer factor of 0.12 per manSv.

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

#### *B.3.3.4 Time Distribution of the Expected Occurrence of Health Effects*

The use of the dose response functions provides the estimate of the total number of health effects expected; however, the details on the expected time of occurrence of these effects has not been addressed. The deterministic health effects that occur after high doses of radiation (accidental releases) will occur in the short-term, but the distribution in time of the stochastic health effects is dependent on two factors:

(1) the continued existence of radionuclides in the environment for years after deposition, and
(2) the latency between exposure and occurrence of the effect.

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

### B.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 B.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.

### *B.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.

### *B.3.4.2 Sea*

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

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

## B.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 B.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.

### B.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:

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

#### *B.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.

#### *B.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.

### B.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.

### B.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).

## B.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 B.6.

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

**Table B.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 |

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.

### B.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.

#### *B.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 $pg/kg_{bw}$·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 B.7. Safety factors reflect the uncertainty involved in extrapolating data between species and also the perceived severity of the effect.

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

In contrast the position adopted by the US Environmental Protection Agency is for an acceptable daily intake about 1000 times lower based on an upper bound risk assessment of

**Table B.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.

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).

### *B.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 B.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 B.8 summarises the results for the plant emitting the highest levels of dioxins considered by HMIP.

**Table B.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 B.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.

**Table B.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 |

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 B.9.

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

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

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 µg/m$^3$

leading to the following ERF implemented in EcoSense:

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

***Threshold assumption:***

WHO 'tolerable daily intake': 10 pg/(kg$_{BW}$·d)

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

*Background:*

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

### B.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 [$\mu g/m^3$] * unit risk factor * Population /70

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

### *B.4.2.1 Cadmium*

**Cancer**

Unit risk factors from

| | |
|---|---|
| ATSDR (1989) | 0.0018 per $\mu g/m^3$ |
| LAI | 0.012 per $\mu g/m^3$ |

**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.

### *B.4.2.2 Mercury*

**Cancer**

Generally not classified as carcinogenic.

**Non-carcinogenic effects**

*Threshold:*

From US-EPA: 0.3 $\mu g/m^3$

**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.

### *B.4.2.3 Arsenic*

**Cancer**

Unit risk factors from

| | |
|---|---|
| WHO (1987) | 0.003 per $\mu g/m^3$ |
| US-EPA (1996) | 0.0002 per $\mu g/m^3$ |
| LAI | 0.004 per $\mu g/m^3$ |

**Non-carcinogenic effects**

*Threshold:*

| | |
|---|---|
| US-EPA | 0.3 $\mu g/(kg_{BW} \cdot d)$ |

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 $\mu g/m^3$

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.

### *B.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.

### *B.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.

## B.5 Occupational Health Issues (Disease and Accidents)

### B.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.

## B.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.

## B.7 Valuation

### B.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.

### B.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 B.10). Both sets vary with discount rate.

### B.7.3 Morbidity

Updated values for morbidity effects are given in Table B.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 B.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 |

**Table B.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. |
| 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. |

**Table B.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. |

### B.7.4 Injuries

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

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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# C AIR POLLUTION EFFECTS ON MATERIALS

## C.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).

## C.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

## C.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).

## C.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 C.1.

**Table C.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 ($\mu g/m^3$)

$O_3$ = ozone concentration ($\mu g/m^3$)

$H^+$ = acidity ($meq/m^2/year$)

$R_H$ = average relative humidity, %

$f_1$ = $1\text{-}\exp[-0.121.R_H/(100\text{-}R_H)]$

$f_2$ = fraction of time relative humidity exceeds 85%

$f_3$ = fraction of time relative humidity exceeds 80%

TOW = fraction of time relative humidity exceeds 80% and temperature >0°C

ML = mass loss ($g/m^2$) after 4 years

MI = mass increase ($g/m^2$) after 4 years

CD = spread of damage from cut after 4 years, mm/year

$Cl^-$ = chloride deposition rate in $mg/m^2/day$

$Cl_{(p)}^-$ = chloride concentration in precipitation (mg/l)

D = dust concentration in $mg/m^2/day$

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

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

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/$m^3$.

### C.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.

### C.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.

### C.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.

### C.4.4 Paint and polymeric materials

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

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

Haynie - carbonate paint:

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

Haynie - silicate paint:

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

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

### C.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.

### *C.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):

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

Butlin - unsheltered zinc (one year):

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

ICP - unsheltered zinc (4 years):

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

ICP - sheltered zinc (4 years):

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

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.

### *C.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

**Table C.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 |

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):

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

ICP - unsheltered aluminium (4 year):

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

## C.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 C.2. The figures given in Table C.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.

## C.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

**Table C.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 |

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 C.3 summarises the damage costs used in this analysis in 1995ECU.

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.

## C.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.

## C.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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# D ANALYSIS OF ECOLOGICAL EFFECTS

## D.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). Assessment of such effects is complicated by the extreme level of site specificity associated with the damage. In most cases in EU Member States local planning regulations should reduce such damage to a negligible level. However, there are inevitably sites where significant ecological resources are affected.

## D.2 Air Pollution Effects on Crops

### D.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 D.1.

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 $\mu g\ m^{-3}$):

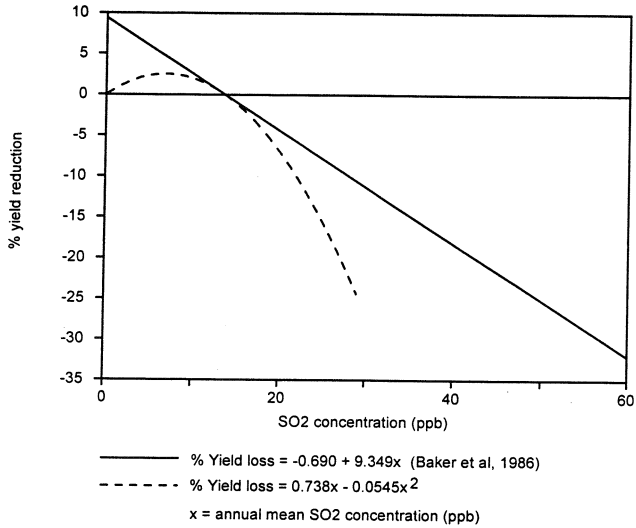



**Figure D.1** Extrapolation of exposure-response functions below the lowest exposure level used experimentally.

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

$r^2 = 0.73$, $p < 0.01$, 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.

### D.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:

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

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 D.1.

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

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

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

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

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 $O_3$.

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;

**Table D.2** Average and standard deviation of yield reduction for species in Table D.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 |

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

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 D.2) which was applied to crops not covered by the functions in Table D.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.

### D.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).

### D.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).

## D.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.

## D.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 D.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.

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.

**Table D.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<br>Scots pine<br>Spruce and/or fir | 1038997 | 10-15<br>(nutrient imbalance) |
| Pine/spruce with oak/birch<br>Pine/spruce with birch<br>Maritime pine | | 10-25<br>(nitrogen saturation) |
| Stone pine<br>Aleppo pine | | 7-20<br>(ground flora changes) |
| Beech<br>Various oaks<br>Cork oak<br>Holm oak | 525642 | 15-20 (nutrient imbalance)<br>10-20 (ground flora changes) |

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

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:

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

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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# E ASSESSMENT OF GLOBAL WARMING DAMAGES

## E.1 Introduction

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

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

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

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

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

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

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

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

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

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

## E.2 Interpretation of Results

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

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

In view of these problems, and in the interests of providing policy makers with good guidance, the task team has sought (though inevitably within limits) to avoid introducing personal bias on issues like discount rate, which could force policy in a particular direction. There is a need for other users of the results, such as energy systems modellers or policy makers, to both understand and pass on information regarding uncertainty, and not to ignore it because of the problems that inevitably arise. The Project team feel so strongly about this that reference

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

## E.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 E.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 E.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.

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

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

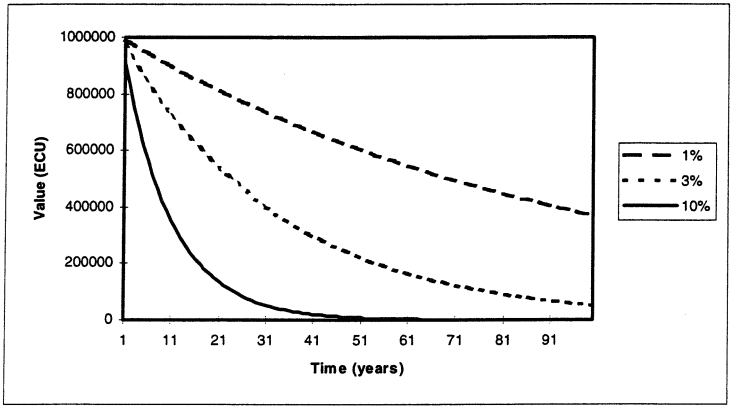



**Figure E.1** Effect of discount rate on present value of damages worth 1 million ECU at the time (from 1 to 100 years in the future) when damage is incurred.

would need to be divided by a factor of $1.6 \times 10^{43}$ 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.

## E.4 Results

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

Data in Table E.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:

**Table E.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

**Table E.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/t$CH_4$ | 520 | 400 |
| Nitrous Oxide, $N_2O$ | ECU/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

- 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 E.2.

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 E.3. Assumptions about higher order effects could affect the results even more.

**Table E.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.

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.

## E.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 E.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/t$CO_2$) 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 E.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)/t$CO_2$** | | |
| Conservative 95% confidence interval | 3.8 | 139 |
| Illustrative restricted range | 18 | 46 |

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# F VALUATION ISSUES

## F.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

## F.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.

## F.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.

## F.4 Transferability of Valuation Data

### F.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.

### F.4.2 Expert Opinion

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

### F.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.

### F.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)

## F.5 Estimation of Uncertain and Risky Phenomena

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

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

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

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

## F.6 Discounting

### F.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.]

### F.6.2 The Discounting Debate from an Environmental Perspective

The relationship between environmental concerns and the social discount rate operates in two directions. In analysing the first, one re-examines the rationale for discounting and the methods of calculating discount rates, paying particular attention to the problem of the environment. In

the second, one looks at particular environmental concerns, and analyses their implications given different discount rates. Beginning with the first, the objections to the arguments for discounting can be presented under five headings:

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

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

### *F.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.

### *F.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.

### *F.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.

### *F.6.2.4 Risk and Uncertainty*

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

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

The risk of death argument is often used as a rationale for the impatience principle itself, the argument being that a preference for consumption now rather than in the future is partly based

on the fact that one may not be alive in the future to enjoy the benefits of ones restraint. The argument against this is that although an individual may be mortal, 'society' is not and so its decisions should not be guided by the same consideration. This is another variant of the view that, in calculating social time preference rates, the pure time preference element (z) may be too high.

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

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

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

### *F.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.

### F.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.

### F.6.4 A Sustainability Approach

The environmental debate has undoubtedly contributed to valuable intellectual soul-searching on the rationale for discounting. But it has not been successful in demonstrating a case for rejecting discounting as such. This Section began by examining the concern over the use of discount rates which reflect pure time preference, but concluded that this concern does not provide a case for

rejecting pure time preference completely. However, it was noted that an abnormally high time preference rate can be generated when incomes are falling and when environmental degradation is taking place. In these circumstances, it is inappropriate to evaluate policies, particularly environmentally relevant ones, with discount rates based on these high rates of time preference.

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

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

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

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

### F.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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# G UNCERTAINTY AND SENSITIVITY ANALYSIS

## G.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.

## G.2 Analysis of Statistical Uncertainty

### G.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 ($\sigma$) of the available data (Figure G.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.

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

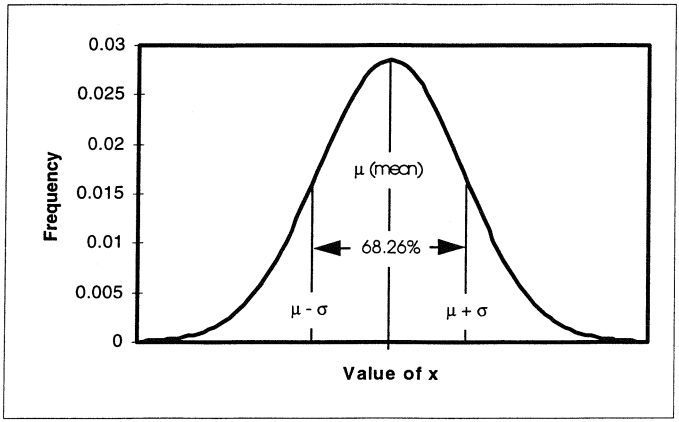



**Figure G.1** Illustration of the normal distribution.

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 ($\mu_g$) and the geometric standard deviation ($\sigma_g$). 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, $\sigma_g$ 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 ($\mu$) and standard deviation ($\sigma$). Figure G.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 ($\mu_g$) and the geometric standard deviation ($\sigma_g$). 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 $\mu_g/\sigma_g$ 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:

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

for the combination of geometric standard deviations. If the median damage has been found to be $\mu_g = 2$ deaths/year, the one $\sigma_g$ 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 G.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.

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.

**Table G.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 $\sigma_g$ |
|---|---|
| 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 $\sigma_g$ | >4.9 |

### G.2.2 Confidence Bands

Estimates of $\sigma_g$ (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 $\sigma_g$ has actually been quantified for a number of impacts (as in Table G.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 $\mu$ by $\sigma_\gamma^2$. The overall ranges represented by the confidence bands are therefore larger than they might at first appear; band C covering four orders of magnitude.

## G.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.

## G.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.

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# H ACRONYMS, UNITS AND CHEMICAL SYMBOLS

## H.1 Acronyms

| | |
|---|---|
| ABB | Asea Brown Boveri |
| AFC | alkaline fuel cell |
| ASU | air separation unit |
| CC | combined cycle |
| CEC | Commission for the European Communities |
| CHP | combined heat and power |
| COPD | chronic obstructive pulmonary disease |
| CORINAIR | CORINE (Coordination d'information environnementale) air emissions inventory |
| CVM | contingent valuation method |
| d | day |
| DGXII | Directorate General XII (European Commission) |
| EC | European Commission |
| ECU | European Currency Unit |
| ERC | Energy Research Corporation |
| ERV | emergency room visits |
| Esc. | Escudos (Portuguese currency) |
| EIA | Environmental Impact Assessment |
| EU | European Union |
| FBC | fluidised bed combustion |
| FDE | Fichtner Development Engineering |
| FfE | Forschungsstelle für Energiewirtschaft |
| FIK | Fachinformationszentrum Karlsruhe |
| GCV | gross calorific value |
| GDP | gross domestic product |
| GESAMP | Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection |
| GHG | greenhouse gas |
| GNP | gross national product |
| GT | gas turbine |
| GUS | |
| GVA | gross value added |
| IFC | International Fuel Cell Cooperation |
| IGCC | integrated gasification combined cycle |
| IKARUS | Instrumente für Klimagas-Reduktionsstrategien |
| IM | Instituto de Meteorologia (Institute for Meteorology ) |

| | |
|---|---|
| INE | Instituto Nacional de Estatística (National Institute for Statistics) |
| IPCC | Intergovernmental Panel on Climate Change |
| IR | Infrared Radiation |
| ISO | International Standardisation Organization |
| ITGCC | integrated tar gasification combined cycle |
| ITL | Italian Lira |
| LTB | low-temperature boiler |
| MBB | Messerschmidt Bölkow Blohm |
| MCFC | molten carbonate fuel cell |
| MECU | million ECU |
| mECU | milli ECU |
| n.a. | not available |
| n.q. | not quantified |
| NCV | net calorific value |
| NCG | non condensable gases |
| NMHC | non-methane hydrocarbon |
| OPEC | Organization of Petroleum Exporting Countries |
| PAFC | phosphoric acid fuel cell |
| PFBC | pressurised fluidised bed combustion |
| $PM_{10}$ | particulate matter under 10 μm diameter |
| RAD | restricted activity days |
| SEDIGAS | Sociedad para el Estudio y Desarollo de la Industria del Gas (Society for the Study and Development of Gas Industry) |
| SOFC | solid oxide fuel cell |
| SPFC | solid polymer fuel cell |
| ST | steam turbine |
| TSP | total suspended particulates |
| UNINOVA | Instituto de Desenvolvimento de Novas Tecnologias |
| UN-ECE | United Nations Economic Commission for Europe |
| USA | United States of America |
| US-EPA | United States Environmental Protection Agency |
| VIK | Verband der Industriellen Energie- und Kraftwirtschaft |
| VSL | Value of Statistical Life |
| WTA | willingness to accept |
| WTP | willingness to pay |
| YOLL | years of life lost |
| yr | year |

## H.2 Units

**Table H.1** SI (Systeme Internationale) Units

| Physical Quantity | Name of Unit | Symbol |
|---|---|---|
| length | metre | m |
| mass | kilogram | kg |
| time | second | s |
| thermodynamic temperature | kelvin | K |
| amount of substance | mole | mol |

**Table H.2** Fractions and multiples

| Fraction or Multiple | Prefix | Symbol |
|---|---|---|
| $10^{-1}$ | deci | d |
| $10^{-2}$ | centi | c |
| $10^{-3}$ | milli | m |
| $10^{-6}$ | micro | μ |
| $10^{3}$ | kilo | k |
| $10^{6}$ | mega | M |
| $10^{9}$ | giga | G |
| $10^{12}$ | tera | T |

**Table H.3** Special names and symbols for certain SI-derived units

| Physical Quantity | Name of SI Unit | Symbol for SI Unit | Definition of Unit |
|---|---|---|---|
| Force | newton | N | kg $m/s^2$ |
| Energy | joule | J | kg $m^2/s^2$ |
| Power | watt | W | kg $m^2/s^3$ (= J/s) |

**Table H.4** Decimal fractions and multiples of SI units having special names

| Physical Quantity | Name of Unit | Symbol for Unit | Definition of Unit |
|---|---|---|---|
| Area | hectare | ha | $10^4$ $m^2$ |
| Energy | gigajoule | GJ | $10^9$ J |
| Energy | terajoule | TJ | $10^{12}$ J |
| Length | micrometre | μm | $10^{-6}$ m |
| Length | millimetre | mm | $10^{-3}$ m |
| Length | centimetre | cm | $10^{-2}$ m |
| Length | kilometre | km | $10^3$ m |
| Power | kilowatt | kW | $10^3$ W |
| Power | megawatt | MW | $10^6$ W |
| Pressure | bar | bar | $10^5$ $N/m^2$ |
| Weight | microgram | μg | $10^{-9}$ kg(= $10^{-6}$ g) |
| Weight | milligram | mg | $10^{-6}$ kg (= $10^{-3}$ g) |
| Weight | gram | g | $10^{-3}$ kg |
| Weight | tonne | t | $10^3$ kg |

**Table H.5** Non-SI units used

| Name of Unit | Definition of Unit |
|---|---|
| °C | degrees Celsius (0°C = 273.15 K) |
| db | decibel |
| GWh | gigawatt-hour |
| h | hour (1 h = 3600 s) |
| kcal | kilo calorie (1 kcal = 4186.8 J) |
| keq | kilo mole equivalent |
| kWh | kilowatt-hour |
| l | litre (1 l = $10^{-3}$ $m^3$) |
| meq | milli mole equivalent |
| $MW_{el}$ | megawatt electric |
| MWh | megawatt-hour |
| $MWh_{el}$ | megawatt-hour electric |
| $Nm^3$ | volume in norm state |
| ppb | parts per billion ($10^9$) by volume |
| ppm | parts per million ($10^6$) by volume |
| rpm | revolutions per minute |
| $TJ_{th}$ | terajoule thermic |
| TWh | terawatt-hour |
| Wh | watt-hour |

## H.3 Chemical Symbols

| | |
|---|---|
| As | arsenic |
| B | boron |
| C | carbon |
| $C_2H_6$ | ethane |
| $C_3H_8$ | propane |
| $C_4H_{10}$ | butane |
| Ca | calcium |
| CaO | lime |
| CFC | chlorofluorocarbon |
| $CH_4$ | methane |
| Cl | Chlorine |
| CO | carbon monoxide |
| $CO_2$ | carbon dioxide |
| $CO_3^{2-}$ | carbonate ion |
| F | fluorine |
| Fe | iron |
| H | hydrogen |
| $H_2$ | hydrogen |
| $H^+$ | hydrogen ion |

| | |
|---|---|
| $H_2O$ | water |
| $H_2S$ | hydrogen sulphide |
| HC | hydrocarbon |
| $HCO_3$ | bicarbonate |
| $HNO_3$ | nitric acid |
| K | potassium |
| Li | lithium |
| Mg | magnesium |
| N | nitrogen |
| Na | sodium |
| $NH_3$ | ammonia |
| $NH_4$ | ammonium |
| $N_2O$ | nitrous oxide |
| $NO_2$ | nitrogen dioxide |
| $NO_x$ | nitrogen oxide |
| $O_3$ | ozone |
| $O^{2-}$ | oxide ion |
| $OH^-$ | hydroxyl ion |
| S | sulphur |
| $SiO_2$ | silicates |
| $SO_2$ | sulphur dioxide |
| $SO_4^{2-}$ | sulphate |
| $SO_x$ | sulphur oxide |
| Sr | strontium |

# EXTERNE

EXTERNALITIES
OF
ENERGY

Vol 9: FUEL CYCLES FOR EMERGING AND END-USE TECHNOLOGIES, TRANSPORT & WASTE

## PART II

# EXTERNAL COSTS OF TRANSPORT

# EXTERNAL COSTS OF TRANSPORT

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# 1. INTRODUCTION

Transport is a source of considerable environmental impacts, mostly representing externalities. The importance of including external costs - to obtain the "true prices" of activities - in economic analysis is agreed, at least at the theoretical level. Recent trends in economic thought, in particular the emphasis on sustainable development and the use of market mechanisms in environmental regulation, have given prominence to environmental externalities.

The aim of this project was to develop a methodology for quantifying energy related environmental externalities of transport, based on a bottom-up approach. The bottom-up approach, which applies a detailed model of emissions and impacts, leads to several important advantages compared to top-down approaches, which are widely used in damage assessment. It is possible to address different fuels, technologies and sites as well as policy measures in a detailed way. These advantages allow to build-up a comprehensive, consistent accounting framework for estimating external costs of transport activities, which is the primary objective of the project.

The starting point was the methodology developed for the electricity generation sector within the ExternE project on fuel cycle analysis. The main feature of this methodology is the modelling of impact pathways. As a consequence of using the existing accounting framework and methodology, the results will be compatible with those of the other ExternE projects. However, there are some important differences between the transport and the electricity sector. In contrast to the fuel cycle study, where stationary high level point source have been considered, the analysis of transport systems has to deal with a multitude of mobile low level sources.

One of the central ideas of this project was to build up a system of "building blocks". A system of basic transport tasks, which can be combined to build complex tasks, were defined and analysed. This approach of using individual transport tasks as "building blocks" allows assessment of marginal and - in a later project phase - aggregate effects, both of which are important in policy. The methodology developed was applied to a series of case studies in Germany, the United Kingdom, the Netherlands, France, Greece and Italy. Environmental externalities due to passenger and goods transport were assessed for a selection of road and rail transport technologies. The results of these case studies provide a basic data set for the building blocks.

For some impact categories (e.g. effects on ecosystems) which currently cannot be translated adequately into external costs, methods were developed in ExternE to analyse the compliance with sustainability indicators.

The quantified and monetised damage (or alternative assessments) of transport activities provides valuable input for decision processes, taking into account the requirements to move towards sustainable transport. These include:

- Internalisation of external costs, e.g. by taxation and other possible internalisation measures.
- Decisions based on cost-benefit analyses, e.g. infrastructure investments, transport measures and policy, research programmes on transport technologies, etc.
- "Green accounting" at the national and European levels.
- Input for E3 models.

# 2. METHODOLOGY AND PRIORITY IMPACT PATHWAYS

The ExternE Project has adopted the 'impact pathway' approach for the assessment of the external impacts and associated costs resulting from the supply and use of energy. The work described in this report builds on the work done in the ExternE Project, which is described in detail in /European Commission 1995 a-f/. The phrase 'impact pathway' simply relates to the sequence of events linking a 'burden' to an 'impact' and subsequent valuation. Emissions and other types of burden such as risk of accident can be quantified and followed through to impact assessment and valuation, whereby for the transport sector in this project the focus lay on impacts due to airborne pollutants and greenhouse gases. The approach thus provides a logical and transparent way of quantifying externalities. However, only recently, through developments in environmental science and economics, and improvements in computing power has it become a realistic proposition.

## 2.1 Guiding Principles in the Development of the ExternE Methodology

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

1. ***Transparency,*** to show precisely how results are calculated, the uncertainty associated with the results and the extent to which the external costs of any fuel chain have been fully quantified. Specific cases are selected in order to provide a clear reference for the date used.
2. ***Consistency,*** of methodology, models and assumptions (e. g. system boundaries, exposure-response functions and valuation of risks to life) to allow valid comparison to be made between different fuel chains and different types of impact within a fuel chain. With the extension of the analysis to the transport sector, a comparison becomes possible even between mobile and stationary energy conversion processes.
3. ***That analysis should be comprehensive,*** we should seek to at least identify all of the effects that may give rise to significant externalities.

In order to comply with these principles, much of our analysis is based in the first instance around the assessment of the effects of individual transport technologies which are closely specified with respect to:

- The technologies used;
- The location of the transport activity and its emissions;
- The type of fuel used;
- The emissions due to production, maintenance and disposal of vehicles and infrastructure.

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

## 2.2 Quantification of impacts

The specific aspects of quantifying impacts due to transport activities are given in section 3 to 9 (dispersion modelling, health impacts, impacts on natural and man-made environment, global warming, up- and downstream processes, economic valuation). The methods which were applied in the same way as for stationary energy conversion processes are discussed in depth in the report on the ExternE methodology /European Commission 1997/. The complexity of the analysis varies greatly between impacts. In some cases externalities can be calculated by multiplying together as few as 3 or 4 parameters. In others it is necessary to use a series of sophisticated models linked to large databases.

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

The most important pollutant transport models used within ExternE relate to the atmospheric dispersion of pollutants. They need to account not only for the physical transport of pollutants by the winds but also for chemical transformation. The dispersion of pollutants that are in effect chemically stable in the region of the emission can be predicted using Gaussian plume models. These models assume source emissions are carried in a straight line by the wind, mixing with the surrounding air both horizontally and vertically to produce pollutant concentrations with a normal (or Gaussian) spatial distribution. For low level emission sources the use of these models is typically constrained to within a distance 30 km of the source.

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

A major problem has so far been the lack of a regional model of ozone formation and transport within fossil-fuel power station plumes that is applicable to the European situation. In consequence a simplified approach has been adopted for assessment of ozone effects /Rabl and Eyre 1997/. In this approach results of the ozone models of EMEP and Harwell were combined with the exposure-response functions and economic valuation for health and agricultural crops recommended by the ExternE Programme /European Commission 1995b/ and updates /Hurley and Donnan 1997/ as well as other publications. Due to the limited geographical resolution of the available ozone modelling data, the estimates are only for average emissions in Europe (EMEP) or at 52° N latitude (Harwell). The resulting damage factors are 1500 ECU/t $NO_2$, 930 ECU/t NMVOC and 130 ECU/t $CH_4$. The nature of the

work necessitated drastic approximations to compensate the lack of data, therefore the results are estimated no better than an order of magnitude.

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

Dose-response functions come in a variety of functional forms, some of which are illustrated in figure 2.1. They may be linear or non-linear and contain thresholds (e. g. critical loads) or not. Those describing effects of various air pollutants on agriculture have proved to be particularly complex, incorporating both positive and negative effects, because of the potential for certain pollutants, e. g. those containing sulphur and nitrogen, to act as fertilisers.

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

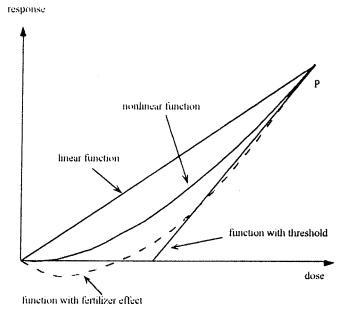



**Figure 2.1:** A variety of possible forms for dose-response functions

The description and implementation of exposure-response relationships is fundamental to the entire ExternE Project. Much of the chapter on health impacts and of the report on methodology /European Commission 1997/ is, accordingly, devoted to assessment of the availability and reliability of these functions.

## 2.3 Economic valuation

The rationale and procedures underlying the economic valuation applied within the ExternE Project are discussed in detail in another ExternE report /European Commission 1997/. The approach followed here is to quantify individual 'willingness to pay' (WTP) for environmental benefit, or 'willingness to accept' (WTA) payment in lieu of environmental harm.

A limited number of goods of interest to this study - crops, building materials, etc. - are directly marketed, and for these valuation data are easy to obtain. However, many of the more important goods of concern are not directly marketed, including human health, ecological systems and non-timber benefits of forests. Alternative techniques have been developed for valuation of such goods, the main ones being hedonic pricing, travel cost methods and contingent valuation.

Hedonic pricing seeks to quantify the worth of one good through variation in the price of another. The best example is probably the economic estimation of people's aversion to noise by reference to variation in house prices between noisy and less noisy environments. Travel cost approaches quantify the worth of something that attracts visitors (ecosystems, landscapes, historic buildings, etc.) by consideration of the money and time spent on visits. Contingent valuation involves setting up a hypothetical market, for example by questionnaire, to elicit the preferences of those interviewed. All of these techniques involve uncertainties, though they have been considerably refined over the years.

The base year for the valuation described in this report is 1995, and all values are referenced to that year. The unit of currency used is the ECU. The exchange rate was approximately 1 ECU to US$ 1.25 in 1995.

The central discount rate used for the study is 3 %, with upper and lower rates of 0 % and 10 % also used to show sensitivity to discount rate. The rationale for the selection of this range and best estimate, and a broader description of issues relating to discounting, was given in an earlier report /European Commission 1995b/.

## 2.4 Assessment of uncertainty

Uncertainty in externality estimates arises from a number of sources, including:

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

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

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

Analogous to the ordinary normal distribution which is characterised by two parameters, the mean and the standard deviation, the lognormal distribution can be characterised by the geometric mean and the geometric standard deviation $\sigma_G$. For this distribution the geometric mean is equal to the median: half of the distribution is above, the other half below the median. The geometric standard has a simple interpretation in terms of the 68 % confidence interval (a

familiar number because for Gaussian distributions 68 % of all values are within one standard deviation of the mean): for a lognormal distribution 68 % of the values are within the interval $[1/\sigma_G, \sigma_G]$. Likewise 95 % are within the interval $[(1/\sigma_G)^2, (\sigma_G)^2]$. Note, however, that these values are centred around the median rather than the mean; the lognormal distribution is not symmetric.

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

In this subject the uncertainties tend to be so large that additive confidence intervals usually do not make sense; instead one should specify 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 normal (for additive processes) or lognormal (for multiplicative processes); thus it is indeed sufficient to specify just two numbers, the mean $\mu$ and standard deviation s for normal distributions, and the geometric mean $\mu_g$ (equivalent to the median) and geometric standard deviation $\sigma_g$ for lognormal distributions. To derive these, each stage of an impact pathway needs to be assessed and standard deviations quantified. The individual deviations for each stage are then combined to give an overall boundary for the parameter under investigation.

After many discussions at various meetings of the ExternE Project a consensus has emerged to use uncertainty labels for each impact, somewhat analogous to the H, M and L confidence levels used in the first series of ExternE reports /European Commission 1995a-f/ but with a more quantitative definition based on geometric standard deviations $\sigma_g$ and confidence intervals of the lognormal distribution. The main reasons for this decision are:
we *can* make meaningful distinctions between different levels of confidence;
providing more definite values of $\sigma_g$ for each impact would imply a false sense of precision, unjustified in view of the need to use subjective judgement to compensate the lack of information about sources of uncertainties and probability distributions;
the labels that we have chosen are intuitive and convey the key message even to readers without training in probability theory, yet they do allow a quantitative interpretation.

The labels are:
**A** = high confidence, 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;

and they can be interpreted in terms of multiplicative confidence intervals: if the cost has been estimated to be $\mu_g$, the probability is approximately 68% that the true value is in the interval $[\mu_g/\sigma_g, \mu_g.\sigma_g]$ and 95% that it is in $[\mu_g/\sigma_g^2, \mu_g.\sigma_g^2]$. (The label "!", used for "extremely

uncertain" in the first reports /European Commission 1995a-f/, has been dropped; we believe that the label C is sufficiently ample, spanning a 95% confidence interval of four orders of magnitude).

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

## 2.5 Priority Impact Pathways

A considerable number of priority impact pathways was identified and addressed (see table 2.1) in the current project the focus lay on the development and application of a methodology for quantifying the external costs due to energy conversion processes. Therefore, only impacts of airborne pollutants and greenhouse gases were assessed in detail. For a comprehensive assessment of transport externalities, of course, impacts due to accidents, noise, etc. should be taken into account.



3)

**Table 2.1:** Priority Impact Pathways considered

| EMISSION/ BURDEN | POLLUTANT | IMPACT |
|---|---|---|
| Benzene | Benzene | Human health |
| 1,3-butadiene | 1,3-butadiene | Human health |
| Carbon monoxide | Carbon monoxide | Human health |
| Combustion particulates | Particulates | Human health |
| Formaldehyde | Formaldehyde | Human health |
| Ethene | Ethene | Human health |
| Lead and compounds | Lead | Human health |
| NOx | NO2 | Human health |
| Poly-aromatic hydrocarbons (PAH) | PAH | Human health |
| $SO_2$ | SO2 | Human health |
| $SO_2$/NOx | Aerosols | Human health |
| Greenhouse gases | Global warming | Various |
| VOC/NOx | Ozone | Human health |
| VOC/NOx | Ozone | Crops |
| VOC/NOx | Ozone | Materials |
| $SO_2$ | $SO_2$ | Crops |
| $SO_2$/NOx | Acidity/nitrogen | Crops |
| $SO_2$/NOx | Acidity/nitrogen | Forests |
| $SO_2$/NOx | Acidity/nitrogen | Ecosystems |
| Crude oil and products in transport | Oil spills and accid. | Ecosystems |
| $SO_2$/NOx | Acidity | Fisheries |
| $SO_2$/NOx | Acidity | Materials |
| Combustion particulates | Particulates | Materials soiling |
| $SO_2$/NOx | Aerosols | Materials soiling |
| Combustion particulates | Particulates | Visibility |
| $SO_2$/NOx | Aerosols | Visibility |

# 3. ATMOSPHERIC TRANSPORT MODELLING

## 3.1 Introduction

The extension of the ExternE Methodology to the transport sector requires the use of new atmospheric dispersion models, compared to the previous phases of the ExternE project. These have been necessary due to the different nature of emissions from transport vehicles, i.e. low level disperse sources, rather than the previous tall, point-source emissions from electricity generation.

The objectives of this task were to identify and demonstrate suitable air dispersion models for the different scales needed within the ExternE Transport Project. This Chapter therefore describes the sub-tasks undertaken in the air dispersion modelling task, and summarises the programme of work undertaken.

## 3.2 Air Dispersion Sub-Tasks

The main sub-tasks undertaken in the air dispersion modelling task were:

- Identification and assessment of existing local air dispersion models;
- Comparison of potential local-scale models;
- Recommendations on the use of the local-scale models;
- Application to the case studies;
- Consideration of regional modelling of transport emissions.

These are discussed in turn below.

### 3.2.1 Identification and Assessment of Existing Air Dispersion Models

A workshop was held at ETSU in May 1996 to evaluate a range of possible air dispersion models for the ExternE Transport project. The workshop concentrated on local scale modelling, though some discussion of regional scale modelling and ozone modelling were included. The local-scale models considered can be categorised into the following groups:

| | |
|---|---|
| Roadside: | MISKAM, CAR; |
| Roadside/Local: | ADMS, CALINE, ROADPOL; |
| Local: | SC, OPS. |

The models were evaluated for their applicability to the project in terms of their type, cost, input data requirements (meteorological data, terrain, etc.), output, range, pollutants covered, chemistry/deposition and if they were validated.

However, the most important criteria in selecting which models to use on the project was their applicability to the impact pathway approach used in ExternE, and most importantly, consistency with the approach for quantifying health impacts. Following confirmation from project health experts, the appropriate output needed from the models was the prediction of average ambient annual concentrations (in the range 5 m - 25 km). The workshop identified a number of promising models which met the criteria for local-scale modelling within the ExternE Transport Project, notably: the ROADPOL, ISC, OPS and ADMS models.

### 3.2.2 Comparison of the Potential Models

The next task was to compare the above models to select the most appropriate for the dispersion modelling activities within the Project It was decided that the models identified should be tested with both simple and complex sets of data to compare their output. A programme of work was undertaken, using common approaches for comparison.

It was decided that provided the output from different models was acceptable, the Project would use the ROADPOL model, as this was the easiest to run for the requirements of the transport project. As part of this task, the ROADPOL, ISC, OPS models were compared under two scenarios. The first was a simple road layout with receptors located within a 1 km box around a line, and also at 1 km intervals away from the vertical line (out to a distance of 50 km); the second a more complicated road layout based on urban data using complex meteorological data. These comparisons were reported in a number of position papers (/Spadaro 1996a/, /Schmid 1996/). An example of the model comparisons are shown in figure 3.1.

As a result of the comparisons, it was concluded the differences between the model outputs was acceptable. It was therefore decided to use the ROADPOL long-term model in calculating annual average concentrations for the local dispersion modelling in the ExternE Transport Project, because of the advantages in the easy use of the model for the transport project needs (i.e. in assessing line sources) and because the ROADPOL model was developed by NTUA (based on an American model code). This allowed the model to be updated specifically to the requirements of the ExternE Transport Project. Indeed, as a result of the model comparisons, a number of alterations were made to ROADPOL, incorporating refinements from the other models assessed. A description of the model is summarised below.

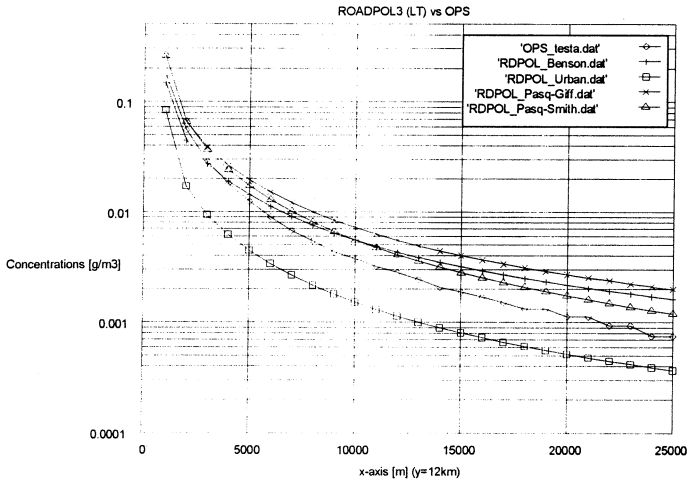



**Figure 3.1:** Comparison of Dispersion Modelling Result from OPS and ROADPOL3 (LT) with Different Dispersion Parameters.

### 3.2.3 The ROADPOL Model

ROADPOL is a Gaussian dispersion model which estimates pollutant concentrations for line sources. The main model used for the ExternE Transport Project was the long-term model (ROADPOL-LT) which calculates annual average concentrations (a short term model was also developed which predicts hourly short term peak concentrations). The model was developed by NTUA. Full details of the model are available in /Vossiniotis et al. 1996a/. In summary:

- The model is user friendly and easy to run, with low run-times;
- The model is a long term line source model, it simulates emissions from a system of line sources and predicts the annual average concentrations of the pollutants in a range of 3 m (kerbside) to 20 km from the emission source;
- The model provides a variety of equations for the proper calculation of the dispersion coefficients and is also able to assess street canyons;
- The parameters considered are the wind speed and direction, the atmospheric stability, the mixing height, the source elevation and the receptor elevation;
- The data required includes:
- Linear source data; the start and end co-ordinates, the height and the emission rate of each road sector,
- Receptor data; the receptor grid co-ordinates and the number of receptors for each direction and the height,

- Meteorological data; the annual frequencies of occurrence for each wind speed, direction , atmospheric stability class and mixing layer height. The model accepts different numbers of wind directions and wind speed class adjusting the input file to the available meteorological data;
- Model data; the definition of the equation calculating the lateral and vertical plume dispersion parameters, and the equation calculating the concentration of pollutants.

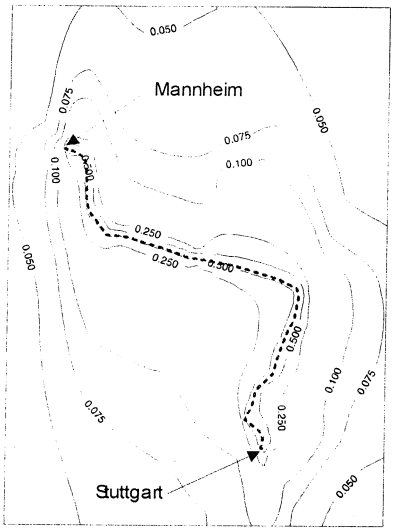



**Figure 3.2:** CO Concentrations (in 1E-12 g/m$^3$) from the Local-Scale Modelling for a Trip by an Articulated Lorry (32t) from Stuttgart to Mannheim (shown by the dotted line).

The model was disseminated to all groups with working instructions and used for the assessment of local air dispersion modelling in the case studies. An example of the typical model data output (converted into a contour map) is shown in figure 3.2.

### 3.2.4 Other Issues Associated with Local-Scale Assessments

There were some additional issues with the local scale assessments which arose during the modelling sub-tasks, and for which it was necessary to undertake further work.

***Local-scale assessment of secondary particulates***

To date, the ExternE Project has considered the health impacts of sulphate and nitrate aerosols (secondary particulates) to be identical to direct particulate emissions. These sulphate and nitrate aerosols are generated from the respective chemical reaction of $NO_x$ and $SO_2$ emissions within the atmosphere. The secondary particles are formed relatively slowly in the atmosphere and have a long atmospheric lifetime. They therefore tend to impact at a regional rather than a local scale. The regional formation and evaluation of these species is already included in the EcoSense model.

However, under certain conditions these secondary particulates can have significant impacts locally (close to the emission source). An example is during summer time smog episodes, when very low wind speeds and bright weather can result in the formation of secondary particulates within urban areas. It was therefore important to assess the potential impacts from these secondary pollutants at a local scale. None of the local-scale models assessed (with the possible exception of the OPS model) included the necessary chemistry to predict the formation of sulphate or nitrate species. Therefore, a simplistic approach for predicting local-scale concentrations was considered (linking the potential rates of formation of these species to the primary emissions of $NO_x$ and $SO_2$).

Although sulphates can form quickly in bright conditions, the approach already quantifies local $SO_2$ impacts and so the error from only assessing sulphate at the regional level is unlikely to be large. The problem is potentially greater for nitrates, because of the very high levels of $NO_x$ emissions from vehicles in urban areas. An assessment was therefore undertaken to investigate the potential impact of local-scale damages from nitrate particulate formation in urban areas /Spadaro 1996b/. The results showed that existing assessment of nitrates at the regional scale within the current approach (using EcoSense) should capture nearly all the potential impact from this pollutant.

It was therefore decided that local secondary particulate formation would not be quantified in this phase of the ExternE Transport Project. However, it is recommended that this issue is addressed in more detail in any subsequent work, particularly for very large urban areas.

*Assessment of peak roadside concentrations*

Previous dispersion modelling activities within the ExternE Project have concentrated on the prediction of annual average concentrations, as this has been the appropriate time resolution for the dose-response functions used in the approach. However, for transport emissions, there is another issue which is important; the assessment of peak concentrations close to roads.

It was therefore necessary to investigate these peak episodes. This work was undertaken in conjunction with a review of the possible health end-points (including the threshold of the effects) from peak concentration exposure. The species considered were $NO_2$, CO and a number of VOCs. A number of model runs were undertaken to look at the roadside pollution concentrations, and to see how significant the possible damages were (bearing in mind that exposure at roadside only tend to be for short time periods). The analysis was also supplemented with measurement data where appropriate. The review confirmed that peak concentrations were unlikely to result in high levels of impacts relative to the existing health damages from annual ambient pollution levels. Nonetheless, it is recommended that this issue is addressed in more detail in any subsequent work, with some specific work addressing the local kerbside concentrations using micro-scale models.

*Assessment of the effect of receptor densities on local-scale concentration estimates*

When undertaking local-scale modelling, the density of the receptor grids at the local scale affects the resulting pollutant concentrations. This can be very important in urban areas, where extremely high population densities occur, as the impact levels can vary significantly according to the receptor grid density chosen. Therefore, a number of test runs of the ROADPOL model were undertaken in order to check the effects of receptor grid density on the resulting concentrations and to make recommendations for a common approach within the project (see /Assimacopoulos and Vossiniotis 1997/). Based on these results, it was recommended that within a distance of 2 km to each side of the road, a dense grid with receptors every 50m should be used (especially if the population density in the vicinity of the road is high), with a coarse grid with receptors every 1000m used beyond this distance.

*Other local-scale modelling issues*

There were a number of other areas, which the team identified as being important, but for which there was insufficient time to address. These included the analysis of pollution dispersion in street canyons and the validation of the air pollution models with measurement data. These issues both need to be considered in future studies to reduce the uncertainty associated with modelling activities.

### 3.2.5 Regional Air Dispersion Assessment

Although the air dispersion modelling task concentrated on the local-scale assessment, some consideration was also made of the dispersion of mobile emission sources at a regional scale.

***The use of the EcoSense Model for transport assessments***

Within the ExternE Transport Project, the EcoSense model has been used for the regional modelling and assessments of impacts. However, for the assessment of transport, a number of updates had to be introduced into the model. The first of these was the extension of the model to include specific transport pollutants, notably CO and carcinogenic hydrocarbons, which have an impact at the regional range. The second was the differentiation between different size fractions of particulate matter ($PM_{10}/PM_{2.5}$). This is important because of the very fine nature of transport particulate emissions. Because of this, primary particulates from transport have a very low deposition velocity and a very long range. The EcoSense regional model (WTM) was thus reconfigured to differentiate between transport and power station particulate emissions.

*Ozone*

In addition to the above work, a review of possible approaches for characterising local, regional and global ozone concentrations was undertaken. This included investigating a number of potential models (from outside the project team). However, because of time constraints, and the modelling effort required for many of these models, a preliminary approach was adopted for quantifying ozone pollution damages of transport pollutants /see Rabl and Eyre 1997/. This method assessed the contribution of transport pollutants to regional and global ozone levels (allocated per tonne of precursor emission), based on results of the EMEP ozone model and the Harwell global ozone model. However, for any future work, the estimation of ozone levels needs to be undertaken in more detail, extending the approach to look at the impact on local ozone concentrations and including detailed model runs for specific case studies to improve the approach at the regional and global level.

## 3.3 Summary

The air dispersion task within the ExternE Transport Project has successfully developed a working approach for modelling pollution dispersion from transport emissions. The approach has specifically addressed the issues of modelling low level disperse-sources from transport emissions and has addressed the issue of modelling pollutants in urban locations, where there can be extremely high receptor densities.

On the basis of a review and comparison of available local dispersion models, the Project has selected and demonstrated the ROADPOL model for quantifying local-scale pollutants. ROADPOL is a Gaussian dispersion model which estimates pollutant concentrations for line sources. It has been primarily used to quantify ambient average annual concentrations within the project.

In addition, the modelling task has updated the regional assessment of transport pollutants, extending the assessment to include carbon monoxide and VOC dispersion at a regional level, and differentiating between the particulate size distribution for transport and power station

emissions (so as to properly evaluate the finer nature and longer range of particulate emissions from transport).

Finally, the task has made assessments of kerbside pollution, local secondary particulate formation and local receptor density and also identified a number of important requirements for future research.

# 4. HEALTH EFFECTS

In this chapter we provide, both for the 'classical' pollutants (particulate matter, PM; ozone, $SO_2$, $NO_2$, CO) and for cancers, the information on exposure-response (E-R) functions for acute and for chronic effects of air pollution, to be used for implementation within the current ExternE Transport project. We list the functions to be used, with brief accompanying background notes.

Briefly, we have surveyed the epidemiological literature in the context of other evidence to form a view on what ambient air pollutants are shown as *associated* with adverse health effects (acute or chronic), and for what specific endpoints; which of these associations may reasonably be interpreted as *causal*; what studies provide a basis for a good set of E-R functions; and how if at all should the E-R functions from individual studies be adapted for use in ExternE.

These are controversial issues. We have focused on studies of appropriate design, with good data, and suitable methods of statistical analysis, including adjustment for confounding factors, checks on model robustness, and where practicable the simultaneous investigation of the effects of multiple pollutants from the general air pollution mixture. We have where necessary re-scaled the chosen E-R functions, if the time-period of daily measurements in the initial study (e.g. 1-hour peak, 8-hour average) did not conform with the units in which incremental pollution is modelled in ExternE Transport.

## 4.1 Representation of particles

There is very substantial epidemiological evidence of adverse acute health effects of particulate air pollution; and strong, but much less widespread, epidemiological evidence of chronic health effects. However, ambient particles are a complex mixture, varying in size and in composition. This leads to E-R functions for particles in terms of total suspended particles (TSP); $PM_{10}$ (inhalable particles); black smoke (BS); finer fractions such as $PM_{2.5}$ or sulphates; and other indices.

The particles of interest in ExternE Transport arise as primary emissions from transport; as sulphates (from $SO_2$); or as nitrates (from $NO_x$). In terms of particle size, particles from all three sources lie principally, if not exclusively, within the finer, $PM_{2.5}$, fraction of $PM_{10}$. There is some evidence that, per unit mass ambient concentration, the relatively fine fractions (e.g. $PM_{2.5}$) are associated with greater risks than $PM_{10}$ generally. Subject to availability of E-R

functions, there is therefore a case for using E-R functions in terms of $PM_{2.5}$ in quantifying the effects of all three sources.

At present, epidemiological studies examining the relationships between particles expressed as $PM_{2.5}$ and health come almost entirely from studies in North America. This does not as yet constitute a comprehensive body of evidence for quantification. Also there is a problem (see below) in transferring E-R functions for acute effects from North America to Europe. Consequently, we have not used the North American functions for $PM_{2.5}$. Instead, we have directly converted from the exposure-response functions for $PM_{10}$ by multiplying by 1.67. This conversion factor is based on /Dockery and Pope 1994/ where it was derived principally from the US experience; there is some indirect European evidence that this is a reasonable factor to use in Europe also.

We have used E-R functions in terms of $PM_{2.5}$ for primary emissions from transport and for sulphates. Nitrate particles are relatively highly soluble and so may be less toxic. Consequently, for nitrates we have used E-R functions in terms of $PM_{10}$.

## 4.2 Ozone

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

## 4.3 $SO_2$

In /European Commission 1995b/, we did not quantify any health effects associated with $SO_2$ as a gas; the evidence at that time was ambiguous. With the APHEA results, an association of $SO_2$ with acute mortality is well-established; and probably one with respiratory hospital admissions also.

It is however still unclear if these associations with $SO_2$ are causal. It maybe that $SO_2$ in these studies is acting as a surrogate for other pollutants, especially fine particles. However, in the APHEA studies the size of the apparent $SO_2$ effect does not depend on the background concentrations of ambient particles; and so, despite the uncertainties, we have use E-R functions for $SO_2$. The estimated impacts are added to the effects of particles and of ozone.

## 4.4 CO

There is relatively little epidemiological evidence concerning CO, and the results are ambiguous. However a few (well-conducted) studies do report positive associations and so provide the basis for E-R functions. On present evidence, we recommend that the functions

for CO and acute hospital admissions for congestive heart failure are used; but that functions for CO and acute mortality are not used.

## 4.5 $NO_2$

In /European Commission 1995b/ we concluded that the apparent acute $NO_2$ effects from some epidemiological studies were best understood not as causal, but as $NO_2$ being a surrogate for some mixture of (traffic-related) pollution.

The recent APHEA results do report positive associations between $NO_2$ and daily mortality or respiratory hospital admissions in several European cities. Consequently, it is possible to propose E-R functions. These however are highly dependent on background particle levels and so the E-R relationships for $NO_2$ are again not used integrally, but for sensitivity analyses only.

## 4.6 Choice of European functions

At the time of working on the ExternE project the meta-analyses from the APHEA study were not available. We relied on published reports of the individual APHEA cities, or other European cities, to form a judgement on the exposure-response functions which would be representative of a European setting.

## 4.7 Transferring E-R functions from North America

Recent epidemiology, especially APHEA, suggests differences in acute effects of air pollutants between Europe and North America. For acute mortality, the estimated effect of particles ($PM_{10}$) is generally lower in Europe while the new estimates for respiratory hospital admissions in Western Europe are about one-half of those estimated previously for ExternE, using US data.

The situation regarding ozone is less consistent. The estimated effect of ozone on acute mortality is much higher in Europe than the US, where evidence is from high-ozone regions only. The relative difference for respiratory hospital admissions is not nearly so great as for mortality.

Consequently, we have sometimes scaled the US or Canadian E-R functions (scaled downwards for particles, upwards for ozone) for use in Europe. The magnitude of appropriate scaling is to some extent arbitrary; our aim has been to improve on no scaling at all, i.e. on an implied scaling factor of one.

## 4.8 Thresholds

For many of these pollutants, there clearly is a threshold *at the individual level*. 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.

For /European Commission 1995b/ the limited quantification of ozone effects did include a threshold. This, however, was principally because of difficulties in ozone modelling, rather than on the basis of epidemiology as such. Overall, the APHEA results do not point to a threshold for the acute effects of ozone; and so we recommend that quantification of *all* health effects for ExternE now be on a 'no-threshold' basis.

## 4.9 Implementation

E-R functions are presented for acute mortality, respiratory hospital admissions, cerebrovascular hospital admissions, cardiovascular hospital admissions, emergency room visits (ERVs), restricted activity days (RADs), acute effects in asthmatics, respiratory symptoms in the general population, chronic mortality, and chronic morbidity.

In terms of implementation for the 'classical' pollutants, the various functions can be used more-or-less directly, by appropriate linkage with incremental pollution and relevant population-at-risk. With regard to chronic mortality, more complex intermediate calculations based on life-table methods are needed.

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

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

## 4.10 Other pollutants, especially carcinogens

As well as the 'classical' pollutants considered above, we need to consider also those other substances released into the atmosphere from transport sources. These include aldehydes (including formaldehyde and acrolein); ethene and ethylene oxide; MTBE (methyl tertiary-butyl ether); VOCs (volatile organic compounds) in general, and benzene and 1,3 butadiene in particular; polycyclic aromatic hydrocarbons (PAHs); diesel exhaust particulates; and lead.

Health effects from exposure to most of these 'new' pollutants from transport sources have not been quantified in previous ExternE studies, apart from the (unpublished) ExternE evaluation of the costs of Energy Recovery from Municipal Waste. Also, the US team has considered the health effects of lead in some detail.

Both the issues, and the available evidence, concerning these 'new' pollutants are markedly different from those concerned with particles, $SO_2$, $NO_x$, CO and ozone. Firstly, with these 'new' pollutants we are dealing principally with chronic, not acute, effects; though for some substances there are identifiable acute effects at ambient outdoor concentrations. Secondly, the basis of epidemiological evidence is generally much weaker: except for lead, there are few good studies of people exposed at ambient outdoor concentrations.

We have evaluated the health effects associated with these 'new' air pollutants arising from transport emissions, with the following specific objectives:

a. For each of the pollutants listed, to identify what health endpoints may be related.
b. To indicate the strength of evidence for each health effect, at concentrations likely to be present in ambient air.
c. In the case of potential carcinogens, or other established health effects, to provide quantitative estimates of unit risks.

We followed the approach of the US Environmental Protection Agency (EPA) and present unit risk factors (URFs) for benzene, benzo[a]pyrene, 1,3 butadiene, and diesel particles, along with formaldehyde and ethylene oxide for sensitivity analyses only.

We have considered the above substances as initiators of cancer. For those who die from cancer as principal cause of death, the average YOLL are estimated as the difference between the expected age of death in the absence of cancer, conditional on having survived to age of diagnosis of cancer and the average age of death among those who die from the cancer. For each type of cancer endpoint identified in the review of transport pollutants, we give average YOLL accounting for age of peak incidence of the cancer in the general population, average latency and average survival rate. These are not definitive scientific assessments. Rather, they are indicative figures only, based on general understanding, to be used as a guide when considering economic assessment, including the effect of discounting.

There is a large body of evidence, including meta-analyses /Schwartz et al. 1994, Needleman et al. 1990/ which suggests that exposure to lead in children has an effect on IQ. These studies are based on the level of lead in blood and so a conversion from lead in air to lead in blood is

necessary to quantify the effect. An E-R function is proposed, using an approximate conversion factor /WHO 1997/ from lead in air to lead in blood, in children.

## 4.11 Concluding remarks

In conclusion, we consider that this quantification framework gives a suitable set of E-R functions for the estimation of health impacts in ExternE Transport. The main omission is not taking account of the acute effects of high, very short-term concentrations of CO. The functions are non-linear and depend on background levels; and so it is not straightforward to derive an implementable set. The issue is being addressed. It will be seen in application that many of the individual pollutant-outcome E-R functions have little impact on final results. This has guided the extent that we have attempted to fine-tune the values provided.

**Table 4.1:** Quantification of human health impacts. The exposure response slope, $f_{er}$, is for Western Europe and has units of cases/(yr-person-$\mu g/m^3$), except for mortality which is expressed as percentage increase per $\mu g/m^3$.

| 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-2 | 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 |
| | 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 |

| Receptor | Impact Category | Reference | Pollutant | $f_{er}$ [1] | Uncertainty rating |
|---|---|---|---|---|---|
| ADULTS (80% of population) | | | | | |
| | Restricted activity days (RAD)[2] | /Ostro 1987/ | $PM_{10}$, | 0.025 | B |
| | | | Nitrates, | 0.025 | B? |
| | | | $PM_{2.5}$, | 0.042 | B |
| | | | Sulphates | 0.042 | B |
| | Minor restricted activity day (MRAD)[3] | /Ostro and Rothschild 1989/ | $O_3$ | 9.76E-3 | B |
| | Chronic bronchitis | /Abbey et al. 1995/ | $PM_{10}$, | 4.9E-5 | A |
| | | | Nitrates, | 4.9E-5 | A? |
| | | | $PM_{2.5}$, | 7.8E-5 | A |
| | | | Sulphates | 7.8E-5 | A |
| | Chronic Mortality (CM) | /Pope et al. 1995/ | $PM_{10}$, | 0.386% | B |
| | | | Nitrates, | 0.386% | B? |
| | | | $PM_{2.5}$, | 0.643% | B |
| | | | Sulphates | 0.643% | B |
| ENTIRE POPULATION | | | | | |
| | Respiratory hospital admissions (RHA) | /Dab et al. 1996/ | $PM_{10}$, | 2.07E-6 | A |
| | | | Nitrates, | 2.07E-6 | A? |
| | | | $PM_{2.5}$, | 3.46E-6 | A |
| | | | Sulphates | 3.46E-6 | A |
| | | /Ponce de Leon 1996/ | $SO_2$ | 2.04E-6 | A |
| | | | $O_3$ | 3.54E-6 | A |
| | Cerebrovascular hospital admissions | /Wordley et al. 1997/ | $PM_{10}$, | 5.04E-6 | B |
| | | | Nitrates, | 5.04E-6 | B? |
| | | | $PM_{2.5}$, | 8.42E-6 | B |
| | | | Sulphates | 8.42E-6 | B |
| | Symptom days | /Krupnick et al/ | $O_3$ | 0.033 | A |
| | Cancer risk estimates | /Pilkington et al. 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 et al. 1996/, /Verhoeff et al. 1996/ | $PM_{10}$, | 0.040% | B |
| | | | Nitrates, | 0.040% | B? |
| | | | $PM_{2.5}$, | 0.068% | B |
| | | | Sulphates | 0.068% | B |
| | | /Anderson et al. 1996/, /Touloumi et al. 1996/ | $SO_2$ | 0.072% | B |
| | | /Sunyer et al. 1996/ | $O_3$ | 0.059% | B |

[1] Sources: /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 4.2:** Human health E-R functions for *sensitivity analysis only* (Western Europe). The slope, $f_{er}$, has units of cases / (yr-person-$\mu g/m^3$), except for mortality expressed as percentage increase per $\mu g/m^3$.

| Receptor | Impact Category | Reference | Pollutant | $f_{er}$[1] | Uncertainty rating |
|---|---|---|---|---|---|
| ELDERLY, 65+ (14% of population) | | | | | |
| | Ischaemic heart disease | /Schwartz and Morris 1995/ | $PM_{10}$, | 1.75E-5 | B |
| | | | Nitrates, | 1.75E-5 | B? |
| | | | $PM_{2.5}$, | 2.92E-5 | B |
| | | | Sulphates | 2.92E-5 | B |
| | | | CO | 4.17E-7 | B |
| ENTIRE POPULATION | | | | | |
| | Respiratory hospital admissions (RHA) | /Ponce de Leon *et al.* 1996/ | $NO_2$ | 1.40E-6 | A? |
| | ERV for COPD | /Sunyer *et al.* 1993/ | Nitrates, $PM_{10}$ | 7.20E-6 | B? |
| | | | Sulphates $PM_{2.5}$ | 1.20E-5 | B? |
| | ERV for asthma | /Schwartz, 1993/ and /Bates. 1990/ | Nitrates, $PM_{10}$ | 6.45E-6 | B? |
| | | | Sulphates $PM_{2.5}$ | 1.08E-5 | B? |
| | | /Cody 1992/ and /Bates 1990/ | $O_3$ | 1.32E-5 | B? |
| | ERV for croup in pre school children | /Schwartz *et al.* 1991/ | Nitrates, $PM_{10}$ | 2.91E-5 | B? |
| | | | Sulphates $PM_{2.5}$ | 4.86E-5 | B? |
| | Cancer risk estimates | /Pilkington *et al.* 1997/ | Formalde-hyde | 1.43E-7 | B? |
| | | | Ethylene oxide | 1.43E-6 | B? |
| | Acute Mortality (AM) | /Touloumi *et al.* 1994/ | CO | 0.00145% | B? |
| | | /Sunyer *et al.* 1996/, /Anderson *et al.* 1996/ | $NO_2$ | 0.034% | B? |

[1] Sources: /European Commission 1995b/ and /Hurley et al. 1997/.

# 5. IMPACTS ON NATURAL AND MAN-MADE ENVIRONMENT

## 5.1 Identification of the impacts of transport activities

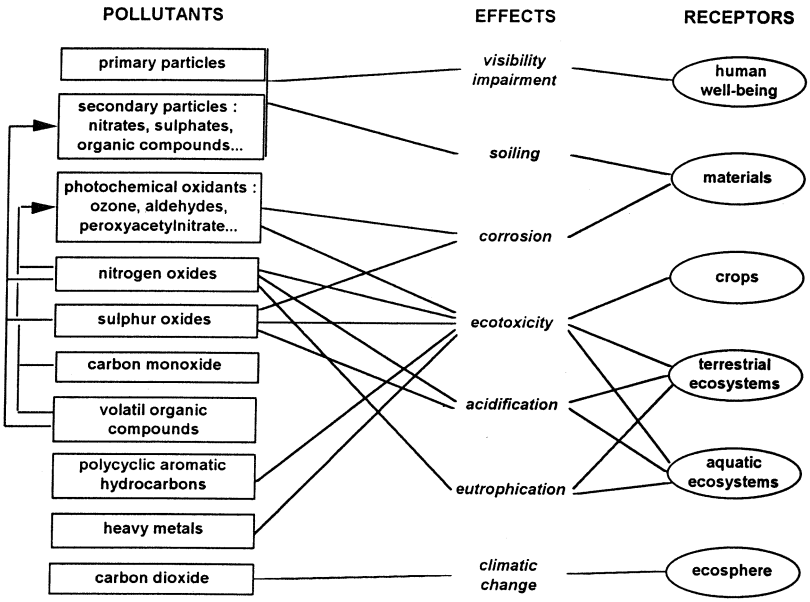



**Figure 5.1:** Some effects on various receptors of transport air pollution

An evaluation by an international panel of experts mandated by the European Environment Agency has highlighted twelve prominent problems concerning the state of the European environment out of a larger unconsolidated list of 56 /Stanners 1995/. A qualitative examination shows that the transport activities contribute to a certain extent to all of those problems. Some quantitative data can give a first evaluation of the importance of that contribution. Figure 5.1 presents a summary of the relationship between pollutants, effects and receptors.

### 5.1.1 Acidification

Acidification of the soils by deposition of sulphur and nitrogen compounds is a well documented and serious problem in large parts of Europe. The part of road transport in the problem of acidification for instance in the Netherlands is globally estimated to about 25 %.

### 5.1.2 Tropospheric ozone and other photochemical oxidants

Ozone concentrations in the lower atmosphere reach levels detrimental to natural vegetation and crops over very large areas in Europe. Ozone and other photochemical oxidants are secondary pollutants produced in the atmosphere by chemical reactions involving especially nitrogen oxides and volatile organic compounds. Road transport, with 63 % of $NO_X$ emissions and 39 % of VOC emissions is clearly the major emitter of ozone precursors.

### 5.1.3 Chemical risk

Transport activities result in the emission in the atmosphere of heavy metals and polynuclear aromatic hydrocarbons (PAHs). Heavy metals are in essence non-degradable and some slowly degradable PAHs are referred to as persistent organic pollutants (POPs). Deposited on ground, they can accumulate in the upper layer of the soil with secondary worrying effects.

For instance, heavy metals that are accumulating in the soil fixed on mineral particles can be ingested with soil by earthworms or grazing animals and enter into food chains. But the reservoir of metals that the soil constitutes may also turn into a source if the binding capacity of the soil is lowered (change in pH due to atmospheric deposition, changes in land use or climatic change...). Then metals released to soil solution can inhibit soil micro-organism activity, be taken up by plant roots, or leached into groundwater, thus affecting drinking water quality. It has been said that such an accumulation could represent a sort of "chemical time bomb".

### 5.1.4 Forest degradation

Forest damages are observed in a large part of Europe. These damages are partly natural (diseases, pests, climatic stresses...) but pollution appears as a crucial cause of weakening. The forest degradation phenomena are complex and transport activities are implicated simultaneously by many pathways (soil acidification, exposure to ozone, surface accumulation of heavy metals, eutrophication by excess nitrogen deposition...).

### 5.1.5 The management of freshwater

Transport activities are a diffuse source of pollution of freshwater by the way of atmospheric deposition. Acidification of waters, especially faced by the Nordic countries, has profound impacts on the biological communities of rivers and lakes.

A short way of surface water chemical contamination by transport pollutants is by runoff. Pollutants such as hydrocarbons, PAHs, heavy metals emitted from exhaust pipe as aerosols are for a significant part deposited on ground near the source, that is on impermeable surfaces as are notably transport infrastructures (highways, airports runways). Following heavy rain episodes these pollutants are sweep along by runoff, contaminating recipient waterbody.

### 5.1.6 Urban stress

About half the energy consumption of road transport take place in the cities ; motor vehicles are the major source of urban air pollution.

As more than two thirds of the European population live in urban areas, the urban transport pollution represents a nuisance for the well-being of people, in terms of noise, odour, visibility impairment, use of land resources, degradation of the urban landscape...

Owing to the conjunction of a high density of emissions of primary pollutants and a high density of receptors, cities are also places of the major impacts on materials.

### 5.1.7 Other environmental problems

Other environmental problems related with the transport activities are very diverse : shortage of resources (Europe with 12.8 % of the world population accounts for 40 % of the energy consumption ; the transport represents 22 % of that energy consumption), major accidents, oil spills...

## 5.2 Quantification of exposure-response functions

Mentioning potential impacts of transport activities shows a large range of diverse, complex and interrelated effects.

### 5.2.1 Complexity of the exposure-response relationship in most cases

In the example given in introduction, that is the yield reduction of crops in function of pollutant concentrations at which these crops are exposed, the exposure-response function observed in experiments appears relatively simple: the function is monotonous and continuous; the response is considered certain, involves short term and can be readily translated in market values. Only a little number of the above mentioned impacts have similar characteristics.

For most of these impacts, the phenomena that may occur in response to a burden are complex. It is generally not obvious to bring to light a variable that sum up the response. The

effects can involve the long term, depend on scenario concerning the context, possibly be irreversible and often are uncertain; the function can be discontinuous (realisation or not of an event).

It is difficult to associate an economic value to such impacts by estimation of the market value of the effects or by enquiries on people willingness to pay. But such effects, that can have an influence on essential ecological equilibrium and the capacity of the earth to sustain human development, are very important to take into account for appreciating the environmental impact of an activity. Much work will be necessary moreover for modelling those effects for decision analysis.

### 5.2.2 Feasibility of a quantification of some effects

Considering the work done precedently in ExternE or elsewhere in the Core Programme, the relative importance of the specific impacts of transport activities and the availability of background scientific information on different effects, some priorities have been chosen in the present phase of the ExternE Transport for deepening the work of modelization of exposure-response functions. Priorities in the field of natural and man-made environment have concerned relatively simple and short term effects: corrosion and soiling of materials, impacts of ozone on crops, and visibility impairments by particles.

Materials and crops have been taken into consideration precedently in ExternE but knowledge in these fields are progressing. Impacts on materials illustrate a case of urban conjunction of a high density of emissions and a high density of receptors. Ozone formation is a burden at a large geographical scale of which first responsibility lies clearly with the transport activities. In the case studies, material degradation due to $SO_2$ and wet acid deposition on zinc, galvanised steel, sandstone, natural stone, limestone, mortar, rendering and paint is quantified. Concerning effects on crops due to $SO_2$, rye, oats, potatoes, sugar beet, wheat, barley were considered. Crop losses due to ozone were taken into account in the damage factors for ozone damages. For materials soiling by primary particles an approach based on /Pons et al. 1995/ was applied, linking cleaning costs for buildings to the particle concentration increment.

Visibility impairment had not been valuated yet in ExternE but was a significant cost item in the American study of coal fuel cycle externalities. It is a category of impacts, like noise and odour, that is in keeping well with the evaluation scheme of the welfare economics and willingness to pay of individuals. A first assessment of visibility impacts was performed but not included in the results. The transferability of the approach and the quality of the data available appear too uncertain.

## 5.3 Directions for new developments

Following the general development of knowledge in environmental sciences, new methodological progress appears obtainable in a number of fields such as :

acidification, eutrophication: An important work on impacts of pollution on natural ecosystems has been made in the framework of the Convention on Long-Range Transboundary Air Pollution. Calculation and mapping system of critical loads for sulphur and nitrogen have been implemented in all Europe, some countries having a very fine level of ecosystem analysis.

*ozone*: New researches are in progress that should give information on exposure-response functions for more crops, with a better taking into account of diverse environmental factors.
river quality: National standards in use could constitute a grid for valuation ; they define scales of quality according to concentrations in water of organic pollutant and other chemicals. Discussions are progressing towards European standards.

*well-being*: As visibility, odours impacts could be taken into consideration. They constitute for people an important aspect of environmental quality and some elements of methodology could be applicable (cf. /Nagy 1991/).

*food chains*: The estimation of human health effects of persistent pollutants require a global environmental analysis. The risk assessment of chemicals in the environment is a discipline whose lessons should be integrated in ExternE.

# 6. IMPACTS OF GLOBAL WARMING

## 6.1 Introduction

Damages due to climate change are one of the most important categories of fossil fuel emission related damages, but also amongst the most uncertain and controversial. The impacts of incremental climate change due to emissions from the European transport sector are truly global in range. Moreover, they will extend over many generations and a very wide range of resources and human activities.

The extent of the impacts, and how they are valued, depends on the level of future emissions of greenhouse gases, the sensitivity of climate to greenhouse gas concentrations, the regional variation in global climate change, the effects on extreme events, the adaptation measures which are put in place and the value systems of the generations affected.

The Inter-Governmental Panel on Climate Change (IPCC) has undertaken detailed analyses of the science of climate change /Houghton et al. 1996/, the physical impacts and potential responses /Watson et al. 1996/ and the socio-economic dimensions /Bruce et al. 1996/. The assessment of damages contained in the last of these /Pearce et al. 1996/ proved one of the most controversial parts of the IPCC Report, because of assumptions made about the value of life in different countries and how this should be used in aggregate damage assessment. There has been subsequent work to address some of the problems raised /Fankhauser et al. 1997/.

Within the context of the ExternE Project the results of the IPCC assessments and subsequent improvements are being handled within two climate damage models - the FUND model at IVM, Amsterdam and the Open Framework at ECU, Oxford. Results will be reported as part of the ExternE Core project later this year. Preliminary results show a wide range of damages, depending on assumptions made about discount rate, equity weighting and higher order damages. The results without higher order damages are similar to those reported in the studies reviewed by the IPCC - a range of $5-$125/tC.

Numerical results reported in the project are therefore based on the IPCC quoted range. But the remainder of this section also includes insights gained and developments made within the ExternE project.

## 6.2 Emissions and Concentrations of Greenhouse Gases

A variety of emissions contribute to climate change. There are basically two categories - direct and indirect greenhouse gases. The direct greenhouse gases are radiatively active within the atmosphere, and therefore contribute directly to global warming. The most important is carbon dioxide, with smaller contributions from methane and nitrous oxide. Indirect greenhouse gases are those which affect, via atmospheric chemistry, the concentrations of other radiatively active species. Potentially important examples are $NO_x$ and VOC via formation of ozone, $SO_2$ via formation of atmospheric aerosols, and a wide range of gases (including CH4, CO and $NO_x$) which react with OH radicals.

It is conventional to report impacts and damages with reference to concentrations or emissions of carbon dioxide. There is therefore a need for some procedure to establish equivalence relations between different greenhouse gases. This is not a trivial issue as greenhouse gases have very different lifetimes in the atmosphere and therefore contribute to global warming over different time periods. A simple approach is to calculate the aggregate contribution to warming over a given period for unit mass emitted at the beginning of the period, compared to the effect of unit emission of carbon dioxide. This is known as the global warming potential (GWP). Because of the differing lifetimes, the GWP is dependent on the time period considered. The IPCC reports GWPs for periods of 20, 100 and 500 years /Schimel et al. 1996/. For the direct greenhouse gases of concern these are shown in table 6.1.

**Table 6.1:** Global Warming Potentials

| Species | Global Warming Potential | | |
|---|---|---|---|
| | 20 years | 100 years | 500 years |
| Carbon dioxide | 1 | 1 | 1 |
| Methane | 56 | 21 | 6.5 |
| Nitrous oxide | 280 | 310 | 170 |

The most applicable of these to climate changes damages is probably the 100 year GWP.

In the ongoing ExternE work on climate change damages, within FUND and the Open Framework, a more exact approach is taken, with atmospheric concentrations of each gas treated separately and decays modelled so that radiative forcing is calculated in each year. Impacts are then calculated and damages summed over the entire period considered. In this way arbitrary equivalence factors are avoided, discounting is enabled and the marginal damages reported separately for each greenhouse gas.

## 6.3 Impact and Damage Assessment

Climate affects a wide range of systems and human activities. There is therefore potentially a huge number of different impacts (including positive ones). However, in general, natural systems and resources are more likely to be directly affected than activities in environments

which are more susceptible to human control. Assessment of impacts of climate change has therefore focused on natural resource sectors and activities which are directly influenced by climate or weather. This approach was used by the IPCC in its assessment of impacts /Watson et al. 1996/. A similar approach is being used in the ExternE Project. The main impact sectors considered are as follows:

- Health
- Sea level rise
- Agriculture
- Water
- Ecosystems and biodiversity
- Extreme events

The approach taken in each of these areas is to use the literature summarised by IPCC and subsequent developments to estimate the impacts at 'benchmark global warming', i.e. the level of climate change when the increased radiative forcing due to anthropogenic emissions is equal to that from double the pre-industrial level of carbon dioxide. Most impact studies to date have been undertaken with this benchmark. This allows comparison of different studies but limits understanding of non-benchmark impacts and damages.

Brief descriptions of impact assessment in the ExternE Project in each sector are as follows

### 6.3.1 Health

Impacts considered are heat and cold stress, vector-borne diseases (particularly malaria), the effects of climate change on pollution related health impacts, and higher order effects due to agricultural and water impacts. For heat stress, cold stress, and vector borne diseases the assessments are based largely on peer-reviewed papers. Models of the effects disaggregated to the regional scale and over time are included in FUND. For pollution effects only the benchmark global effect of ozone is estimated using a geographical analogue to estimate temperature effects. For higher order damages, the impacts are very uncertain, but potentially very large, being very dependent on the underlying socio-economic conditions.

### 6.3.2 Sea level rise

Analysis of the effect of sea level rise on coastal systems has a long tradition in the study of the impact of climate change. Major effects include the inundation of dry- and wetlands, increased severity of sea surges, and intrusion of salt water. To date, only inundation has been satisfactorily quantified. The main problem is quantifying human adaptation: additional coastal protection will reduce the amount of dryland lost (and may well increase the amount of wetland lost), but this depends on insufficiently known human behaviour. Damage estimates are based on the IPCC Common Methodology or its predecessor, the Global Vulnerability Assessment. Both methods use rules of thumb to decide on the extent of protection, in the former case depending on population densities and in the latter on per capita income levels.

More refined behavioural models reveal that these rules of thumb are very crude, but these models have only been applied in fairly limited case studies.

### 6.3.3 Agriculture

Agricultural impacts are treated in different ways in the two models because of the different model structure. In the Open Framework, a simple climate-agriculture model based on ESCAPE links climate change to the value of agricultural production at the national level. In FUND, the regional sensitivity of agricultural production to global benchmark climate change is taken from a review of the literature. The former approach allows a greater geographical sensitivity. The latter, although more geographically aggregated, allows a better representation of the dynamics of climate change. In both cases, value is determined with reference to the agricultural component of GDP rather than specific commodities. A more sophisticated understanding of climate change and agriculture would take into account the effect of world trade and price feedback. The key uncertainties are the pace of technological innovation in agriculture and the linkages between agriculture, regional economies, ecosystems and human health. Large costs are only realised if technology stagnates, agriculture continues to dominate developing economies, expansion of agriculture (rather than intensification) to meet world demand destroys valuable ecosystems, or the failure of agriculture in some regions leads to widespread poverty and famine with significant increased mortality. At least as likely is for agriculture to be a net beneficiary of climate change and for the adverse effects to be within the range of adaptation assuming reasonable levels of world trade.

### 6.3.4 Water

Water supply and demand damages are not incorporated in FUND. The Open Framework links a simple water balance model with a price/demand economic calculations on an national basis. The results show significant damages, with regional disparities between the winners and losers. Water is notably different from agriculture, in that it is not traded at a global level. Thus adaptation will be managed at the local level. On the other hand, water is more likely to be a source of conflict and this could lead to quite large damages in some regions. The key uncertainty is the effect of enhanced $CO_2$ levels on water use efficiency. If this is significant, the impact of climate change on water resources could be quite small. Small changes in rainfall patterns should be well within the adaptive ability of water utilities, but adaptation would entail some costs, in new reservoirs and managing demand more effectively.

### 6.3.5 Ecosystems and biodiversity

All ecosystems will be affected by climate change. The seriousness of the threat depends on the ability of the system to adapt and/or to translocate to more climatically suitable regions. The ecosystems most likely to be seriously affected are forests, coastal systems at risk from sea level rise, and other isolated ecosystems such as those on islands, mountain peaks and in the polar regions. The total economic value of ecosystem damage is the sum of the use value and the existence value. The former measures the input as a resource makes to economic activity and the latter the aesthetic aspects. Existence values are measured using the contingent

valuation method (CVM), which raises a large number of concerns, so that results of CVM studies on the natural world are not wholly convincing. Damage to ecosystems and their diversity is increasingly recognised to pose unquantifiable threats to the ecosystems functions on which human societies rely and, through loss of biodiversity, to system stability and resilience. These have been considered far less than aesthetics in the climate change valuation literature, but are potentially more important. It is concluded that monetary valuation does not provide a comprehensive assessment of ecosystem value.

### 6.3.6 Extreme weather events

Extreme weather events are generally believed to be major causes of impacts of climate change and major drivers of adaptation. Yet, most attention has been paid to smooth changes in the means of weather variables. Exceptions are the impact of tropical cyclones. Impacts were initially thought to be large and negative, but reanalyses of the physical links lead to much less concern. River floods are being added. Case studies point at a sensitivity to precipitation changes which may be substantial and adverse. Human adaptation in the flood plain is just as important; it can exacerbate or reduce impacts. Only one study of extra-tropical storms has been reported; it concludes that the exposure of Western Europe to storms is so low that a possible doubling or tripling would do little damage. Droughts have not yet been taken up in marginal damage estimates of greenhouse gas emissions, although understanding the phenomenon is increasing.

## 6.4 Damage Values

The global warming work within the ExternE Core project will provide an approach, consistent with the ExternE methodology, for global warming damages and estimates of the range later this year. In the interim, we use of the range of possible damage values reported by the IPCC /Pearce et al. 1996/. This is $5-125/tC, which is approximately equal to 4-100 ECU/tC, or 1-30 ECU/t $CO_2$.

For methane and nitrous oxide, it is appropriate to use the same range multiplied by the 100 year GWP, i.e. 21 for methane and 310 for nitrous oxide.

It is worth noting a few points about this range:

- A major sensitivity is discount rate. Recent ExternE work indicates that for many key damages (including human mortality) an effective discount rate close to zero is appropriate. This corresponds to the upper end of the range, although the majority of commonly quoted studies use a higher rate and therefore calculate a lower value.
- The results are usually taken from aggregates of regional damages, where the damages in developing countries are based on extrapolations from OECD regions to which most detailed impact and damage studies relate. Moreover, they are simple aggregates with no equity weighting implying no declining marginal utility of income.

- The IPCC survey excludes studies which find high values for higher order damages like famine and war (e.g. /Hohmeyer and Gärtner 1992/; /Ferguson 1994/; /Kuemmel and Sørensen 1997/). Preliminary results in the ExternE project indicate that these higher order effects, although very uncertain, could be significantly larger than other damages.
- The results are very uncertain. The reported range is from 'best guesses' of various studies, not a genuine uncertainty range. Indeed the IPCC itself issues the following caution: 'This range of estimates does not represent the full range of uncertainty..... It must be emphasised that the social cost estimates have a wide range of uncertainty because of limited knowledge of impacts, uncertain future technological and socio-economic developments, and the possibility of catastrophic events or surprises.'

It must also be noted that the studies surveyed by IPCC are often inconsistent with ExternE in some ways, for example in the treatment of health valuation. However, preliminary results from the ExternE project indicate that a similar range to that reported by the IPCC will be obtained for damages without higher order effects. It is therefore used in the absence of more reliable values.

## 6.5 Uncertainty

It is clear that assessment of climate change impacts and damages is very complex and currently rather poorly developed. Even where different studies have reached similar values, caution is required, because in some cases it is the similarity of assumptions which drives the results, and the assumptions are not necessarily well founded or correct. Wherever a wide range of reasonable assumptions about the future and how to evaluate it have been taken into account, it has been found that the damages of climate change are very uncertain.

In particular, the results are very sensitive to some key assumptions, which are not made sufficiently transparent in all studies /Eyre 1997/. These sensitivities include:

- reasonable values for the discount rate,
- some impact categories which are very sensitive to assumptions made about the potential for higher order damage (health effects, migration, conflict etc.),
- different assumptions about the form of the damage function and the dependence of damages on the level or rate of climate change,
- different methods for aggregating individual damages (equity weighting), and
- for the important category of ecosystem impacts, major practical and theoretical problems in the methodology.

Different, reasonable assumptions about these issues can each alter damage estimates by an order of magnitude. Moreover, they interact in some complex ways, so that they can not only alter the overall damage estimate, but also lead to different conclusions about which damages categories are important.

These conclusions are not a reason to reject impact valuation, rather to ensure that all future damage estimates are carefully qualified by an assessment of what assumptions have been made and what the effects of different assumptions might be.

## 6.6 Conclusions

The ExternE Project work on climate change damages is incomplete.

In the absence of ExternE values, it is recommended that the damage range quoted by the IPCC should be used, that is $5-125/tC, equivalent to 1-30 ECU/t $CO_2$. For other greenhouse gases, this numerical range should be multiplied by the 100 year GWP - 21 and 310 for methane and nitrous oxide respectively.

Preliminary data from the ExternE Core Project indicates results broadly consistent with this range, if catastrophic and higher order damages are neglected. It should be noted that the range probably fails to encompass the full range of uncertainty. The values quoted by the IPCC are largely dependent on discount rate (with higher rates giving lower damages). Not only is the possibility of catastrophic or large higher order damages neglected, but the full range of climatic and socio-economic scenarios is not covered.

# 7. UP- AND DOWNSTREAM PROCESSES

One of the central features of the ExternE Methodology is that the impacts from all stages of the fuel cycle are considered. Thus, for transport the impacts from fuel production, vehicle production and the supply of infrastructure must be considered in addition to vehicle use. It has to be verified if the greatest impacts are likely to arise from the operation of vehicles, and how important the burdens arising from all the other life-cycle stages are.

**Table 7.1:** Up- and downstream processes for transport vehicles

| Major sections | Sub-sections |
|---|---|
| Fuel Production | Raw material extraction |
| | Feed stock transportation |
| | Process plant construction |
| | Processing |
| | Fuel distribution |
| | Vehicle refuelling |
| Vehicle Manufacture | Raw material extraction |
| | Transportation |
| | Processing |
| | Factory construction |
| | Component manufacture |
| | Component transportation |
| | Assembly |
| | Vehicle distribution |
| | Disposal / recycling |
| Vehicle Use | Emissions factors |
| Vehicle Maintenance and Support | Component manufacture |
| | Distribution |
| | Facility construction & maintenance |
| Infrastructure Construction and Maintenance | Raw material extraction |
| | Transportation |
| | Processing |
| | Earth work |
| | Road crust |
| | Bridge and tunnel construction and maintenance |
| | Supplies |
| | Service |

At first the life cycle stages that should be considered were reviewed as well as existing studies quantifying energy use and emissions of these stages. Furthermore, the impacts due to crude oil transportation in the Mediterranean Sea were looked at. From existing studies a set of average values (applicable to European transport activities) was produced for use when country-specific data was not accessible.

## 7.1 Existing Studies

Table 7.1 presents the life-cycle stages that have been found most important in the transport sector. Due to the problems as regards data availability and data complexity only few studies to date consider construction and maintenance of infrastructure. This appears to be an important shortcoming, as the studies which analysed these process steps suggest that the emissions are far from being neglectable, particularly for rail transport.

A study which covers the full range of process steps for a broad range of technologies was published by /Maibach et al. 1995/. The main focus of the study lies on transport in Switzerland, but as many of the up- and downstream processes take place in Europe, Western Europe is covered as well. Following, the parts of the study relevant for ExternE Transport are briefly described.

The analysis of the various process steps comprises:

**Vehicle production:**
Material and energy use for production and processing of base materials and energy processes. No fully detailed process chain analysis, but use of general assumptions. Production and disposal of means of production (buildings, productive facilities etc.) are not taken into account. For electricity use the UCPTE-mix (Union pour la co-ordination de la production et du transport de l'électricité - the European electricity supply grid) is assumed.
**Vehicle maintenance:**
Material and energy use for components; specific emissions (e.g. VOC due to varnishing); operation of garages (electricity and fuel oil).
**Vehicle disposal:**
Not-recyclable share of scrapped vehicles according to current practice in Switzerland. Energy use is not considered, as energy use per vkm is assumed negligible compared to vehicle use.
**Infrastructure construction:**
Construction of roads, railway lines, etc. and essential facilities.
**Infrastructure operation/maintenance:**
De-icing salt, electricity use for tunnels, etc.
**Infrastructure disposal:**
Dismantling, material transport, material dump.
**Fuel production**
**Prospection, exploration, processing and transport of fuels.**

Table 7.2 shows the reference area underlying the analysis in /Maibach et al. 1995/.

**Table 7.2:** Reference area for process steps by vehicle technologies in /Maibach et al. 1995/

| Technology | Vehicle | | | Infrastructure | | |
|---|---|---|---|---|---|---|
| | Production | Operation/ Maintenance | Disposal | Construction | Maintenance | Disposal |
| Car | D | D | CH | EU | EU | EU |
| Van | CH/EU | D | CH | EU | EU | EU |
| Lorry | CH/EU | D | CH | EU | EU | EU |
| Coach | CH | CH | CH | CH | CH | CH |
| Bus | CH | CH | CH | CH | CH | CH |
| Tram | CH | CH/EU | CH | CH | CH | CH |
| Inter-city train | CH | CH | CH | CH | CH | CH |
| High speed train | D | D | CH | D | D | D |
| Short-distance train | CH | CH | CH | CH | CH | CH |
| Goods Train | CH | CH | CH | CH | CH | CH |

EU = Reference European average; D = Reference Germany; CH = Reference Switzerland
CH/EU = Reference Switzerland with UCPTE electricity mix

Among other burdens, following emissions of airborne pollutants are quantified: $CH_4$ (methane), $N_2O$ (nitrous oxide), $CO_2$ (carbon dioxide), CO (carbon monoxide), NMVOC (non-methane volatile organic compounds), $SO_x$ (sulphur oxides), $NO_x$ (nitrogen oxides) and particles. Methodology and data are mainly based on work carried out by /Frischknecht et al. 1994/ analysing energy systems. The year 1993 was selected as reference year.

Apart from /Maibach et al. 1995/, only little data is available on the process step infrastructure. Figure 7.1 compares the emissions due to infrastructure use per vehicle km for a lorry with 40 t gross vehicle weight. The shaded bars give the values from /Maibach et al. 1995/, the bright bars show the results of a study carried out by /ISV et al. 1997/. In the latter study, specific goods transport relations in the federal state of Baden-Württemberg (Germany) were examined in a detailed process chain analysis. Even though, according to figure 7.1 there are considerable differences, the emissions per vkm are within the same order of magnitude - except for VOC. $SO_x$ emissions were not quantified by /ISV et al. 1997/.

In view of the multitude of parameters involved - as there are assumptions on material input, upstream processes for infrastructure construction, life time of infrastructure facilities, etc. - the differences appear tolerable. The characteristics of a specific site can make a big difference, as illustrated in /ISV et al. 1997/ for road infrastructure: the total energy use quantified for construction, operation and maintenance per road-km and year may differ by a factor of two within the same road category. Another source of difference may be the criteria used to apportion the total emissions to the vehicle categories. /Maibach et al. 1995/ use the share of gross tonne kilometres run on a road, whereas /ISV et al. 1997/ apply factors taking into

account axle load and vehicle speed (wear and tear of the road) as well as effects of vehicle size and speed on the use of the road capacity.

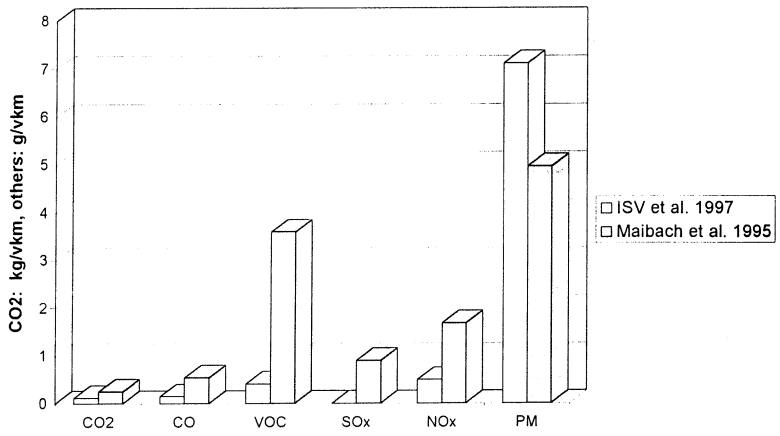



**Figure 7.1:** Emissions due to infrastructure use by a lorry of 40 t gross vehicle weight

Furthermore, an additional review has been made of alternative fuels for road transport vehicles /Collings 1996/. These include petrol, diesel and other alternative fuels including LPG, CNG, Biomethanol, Bioethanol, Biodiesel and electric vehicles. This work concentrates on the three major life-cycle stages - production of the motor vehicle, production of the fuel used to power the vehicle, and the use of the vehicle on the road. The best available data for the life-cycle emissions were collated from a range of studies (/ETSU 1996/; /Rijkeboer and Hendriksen 1993/; /DeLuchi 1991/; /Agnetun et al. 1993/; /Johansson et al. 1992/). There is a considerable variation in some of the emissions factors between the different studies, even though they have all investigated the full fuel production life-cycle. It should be noted that for electrically powered and hydrogen fuelled vehicles, and for vehicle manufacture, the life-cycle emissions are heavily dependent upon the method of electricity generation used. Clearly, this varies considerably from country to country.

Figure 7.2 shows the emissions due to car production of /Collings 1996/ compared to the respective values from /Maibach et al. 1995/. The figures compare well, except for particulate matter. This may be caused by a different level of detail of the process chain analysed.

The aggregated life-cycle emissions for fuel production, vehicle production and in-service use for petrol-powered vehicles for a medium sized family car (g per vehicle kilometre) are shown in table 7.3. The total aggregated life-cycle emissions for vehicles powered by a range of fuels measured in g per vehicle km are presented in table 8.4.

**Table 7.3:** Life-cycle emissions for a medium sized family car /Collings 1996/.

| g/vkm | $CO_2$ | CO | $NO_x$ | VOC* | $SO_2$ | PM | $N_2O$ |
|---|---|---|---|---|---|---|---|
| Fuel Production | 47.0 | 0.061 | 0.174 | 0.388 | 0.185 | 0.011 | 0.0066 |
| Vehicle Production | 54.5 | 0.021 | 0.160 | 0.105 | 0.493 | 0.016 | 0.0007 |
| Vehicle Use | 186.3 | 3.371 | 0.224 | 0.299 | 0.020 | 0.005 | 0.0538 |
| Total | 287.8 | 3.453 | 0.558 | 0.792 | 0.699 | 0.032 | 0.0611 |
| % of total emissions | | | | | | | |
| Fuel Production | 16.3 | 1.8 | 31.1 | 49.1 | 26.5 | 35.7 | 10.9 |
| Vehicle Production | 18.9 | 0.6 | 28.7 | 13.2 | 70.6 | 48.8 | 1.2 |
| Vehicle Use | 64.7 | 97.6 | 40.2 | 37.7 | 2.9 | 15.5 | 88.0 |

* including VOC emissions from paint and adhesive operations.

**7.4:** Total life-cycle emissions (g/vkm) for a car powered by different fuels /Collings 1996/.

| Total Emissions | $CO_2$ | CO | $NO_x$ | VOC | $SO_2$ | PM | $N_2O$ |
|---|---|---|---|---|---|---|---|
| Petrol | 287.8 | 3.453 | 0.558 | 0.792 | 0.699 | 0.032 | 0.0611 |
| Diesel | 227.1 | 0.489 | 0.981 | 0.384 | 0.702 | 0.131 | 0.005 |
| LPG | 239.0 | 3.889 | 0.482 | 0.443 | 0.649 | 0.027 | 0.505 |
| CNG | 242.0 | 0.863 | 0.457 | 1.137 | 0.575 | 0.022 | 0.815 |
| Methanol | 233.7 | 3.292 | 0.729 | 0.914 | 0.549 | 0.023 | 0.038 |
| Biomethanol | 292.0 | 3.419 | 0.784 | 0.597 | 0.646 | 0.039 | 0.046 |
| Electricity | 228.1 | 0.068 | 0.520 | 0.451 | 1.005 | 0.040 | 0.008 |

In a screening study the impacts of crude oil transportation by sea were studied, with an emphasis on the crossing of the Mediterranean /Vossiniotis et al. 1996b/. The process steps of crude oil transport are:

- Loading crude oil;
- Travelling towards destination;
- Unloading;
- Cleaning of tanks;
- Return journey.

The priority impacts of the process have been specified to be:
1. Impacts on natural environment due to operational and accidental discharges of oil;
2. Impacts on financial and social activities due to operational and accidental discharges of oil;
3. Occupational health effects.

Data on crude oil losses during sea transport and accidents were collected. A total amount of 330,000 tons of crude oil is estimated to be discharged every year in the Mediterranean. Impacts due to crude oil losses have been assessed based on the compensation costs of the

Amoco Cadiz spill. Due to the high uncertainties of this estimate and the issue of transferability, it was not used within this study.

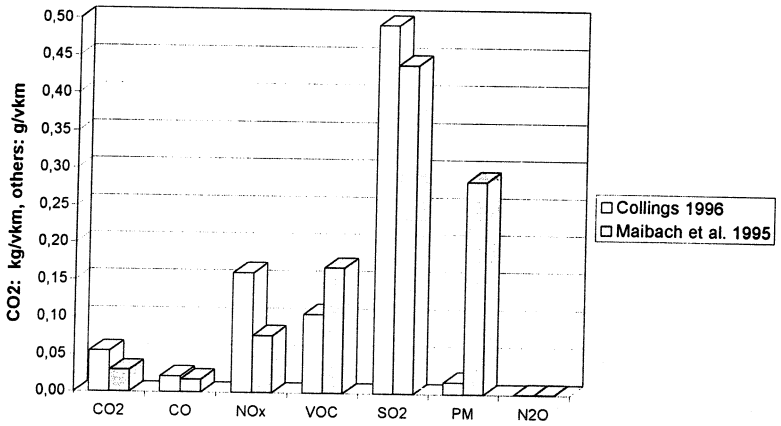



**Figure 7.2:** Comparison of emissions due to the production of a medium sized family car

## 7.2 Assessment of up- and downstream processes in ExternE Transport

### 7.2.1 Average Emissions

As a detailed assessment of up- and downstream processes was beyond the scope of the project, estimates had to be based on existing studies. It was decided to use /Maibach et al. 1995/ as data source for several reasons. First of all it covers the full range of process steps with a common methodology. In addition, data are provided for all technologies considered in the case studies. As shown in table 7.2 above, the reference area was Switzerland, Germany and the EU (i.e. Western Europe), so the estimates should be compatible with all case studies. To conclude, /Maibach et al. 1995/ was the only study providing a complete set of data to be used for a first assessment of up- and downstream impacts.

As figure 7.1 shows, the particle emissions given in /Maibach et al. 1995/ and other studies are considerable. Only a small fraction of the particles quantified stems from combustion processes and therefore is relevant for impact assessment. The major part is coarse dust produced during raw material extraction and processing and is not relevant for the health effects quantified in this study. To give an example, 90% of total particles emitted in relation with the high speed train infrastructure are released during limestone extraction as coarse dust.

To avoid overestimation of impacts, the fraction of coarse particles for the process steps was checked and corrected accordingly. For instance, only 1% of the particles of infrastructure processes for high speed trains are combustion particles, for road infrastructure the respective fraction is 5% and for car production it is 10%.

### 7.2.2 Assessment of Damages

With the results of the ExternE programme real damage cost estimates are available that can be integrated into LCA, in addition or as an alternative to the potential impact indices used until now. Even though the site dependence of damages has been considered an obstacle, first estimates can be made based on plausible assumptions. The impacts from the use of vehicles are assessed in detail, as the emissions close to the population lead to considerable ambient concentrations. As the emissions from up- and downstream processes in general do not occur in densely populated areas they were treated as diffuse background as a first approximation. Hence, damage factors per tonne of pollutant emitted, which can be derived from impact assessment on the European scale, can be used to obtain a first estimate of the damages due to up- and downstream processes.

# 8. ECONOMIC VALUATION

## 8.1 Introduction

We used the Damages Function (DF) approach developed in the previous phase of the ExternE project /European Commission 1995b/. The only difference in the "economic valuation" - step of the DF approach for emissions from transport compared to fuel cycles, is that economic values for new types of impacts had to be derived. We also adjusted some of the economic values used in the previous phases of ExternE based on a new literature survey and the recent work on mortality valuation done within the *ExternE Maintenance, Improvement, Extension and Application of the ExternE Accounting Framework* - project (ExternE Core project). All values are expressed in 1995-prices (through a simple adjustment of values with the consumer price index for the EU).

The analysis is based mainly on a review and evaluation of the existing studies on economic valuation of environmental and health impacts, with emphasis on European valuation studies. In addition, the alternative approach of Quality Adjusted Life Years (QUALY) to evaluation of health impacts has been reviewed and evaluated based on the existing theoretical and empirical studies in this field.

## 8.2 The QUALY approach to morbidity and mortality

The theory of Quality Adjusted Life Years (QUALY) is used to compare the quality of life of different health states, combining morbidity and mortality in a single outcome indicator. Survey techniques are used to elicit peoples evaluation of different health states. Different health states are directly or indirectly assigned a value from 0 to 1, where 1 is full health and 0 is death. A value of 0.25 means that one life year in this health state is equal to 0.25 life years at full health. In other words; one life year with this health condition represents a loss of 0.75 life years at full health. These weights used to reflect values of the quality of life can be seen as utilities as they are obtained by asking people (i.e. the general public or specific groups of people e.g. having experienced the health state in question) questions which allows the analysts to derive their implied preferences.

Although the large literature on QUALY, both theoretical and empirical work shows that it is a potentially powerful tool, the problems are numerous and have been well documented /Loomes and McKenzie 1989/.

/Nord 1992/ describes five techniques used to measure the quality of life of a certain health state. These methods can give very different answers. He points out weaknesses of all five methods, but he also concludes that the Time Trade-Off method is the best taken into consideration weaknesses and strengths of all methods.. A good test of the reliability and validity of the methods is probably not possible, and has not been performed (Nord op. cit.).

Which ever method is used, a problem with QUALYs is that the utility weights for a given condition generally are multiplied by the time spent in that condition. The assumption is that utilities are independent of the time spent in that state. Yet people tend to rate the disutility of a particular condition less highly if it involves only a short period of suffering. It is also necessary to assume that utility in a given health state is independent of the previous state of health and the states of health expected in the future - all of which have been shown to be inconsistent with people's actual preferences /Loomes and McKenzie 1989/. Also, QUALYs, in the way they are calculated, usually assumes risk neutrality while people may he risk averse.

Due to the lack of health endpoints in exposure-response functions quantified in terms of QUALY, and the weaknesses of methods for eliciting QUALY values for different health states, we recommend that the QUALY approach is not used in the ExternE projects. However, some of the work done using the QUALY approach can contribute to shed light on e.g. age dependency of VSL and how VSL varies with the cause of death and whether it is acute or chronic mortality.

## 8.3 Mortality

Starting point for mortality valuation was the Value of a Statistical Life (VSL) of 3.1 million ECU (i.e. 2.6 Million ECU in 1990-prices adjusted with the consumer price index to 1995-prices). This estimate is based on a thorough review of the existing valuation literature on mortality: see /European Commission 1995b/. The available estimates of willingness-to-pay (WTP) to prevent incremental changes in risks of death are based on situations where individuals are observed making trade-offs between risks of death and income.. Most of these studies have estimated wage premiums associated with different levels of on-the-job risks of fatal accidents (i.e. hedonic wage (HW) studies). Additionally, some contingent valuation (CV) surveys have been conducted in which respondents have been asked about their WTP to reduce e.g. the risk of fatal accidents at work or in road traffic.

For the most part, available WTP estimates are for risks of accidental death in circumstances where individuals are voluntarily exposed to risks (e.g. choosing a job or driving a car). The estimates are also drawn mainly from studies of working-age adults. Some potentially important differences exist between the context of these available estimates and the mainly environmental health risks being evaluated in the ExternE projects. Most of the valuation studies value current risks of accidental death, whereas many of the environmental health risks are likely to cause death after a latency period (e.g. cancer and chronic respiratory illnesses). There are also some evidence that people are more concerned about avoiding some kinds of deaths, e.g. cancer deaths, than an unforeseen instant death (from motor vehicles) often

estimated in the VSL-literature /Jones-Lee et al. 1985: 68/. Also, many of these deaths are due to involuntary exposure to air and water pollutants. In some cases the environmental risks also falls disproportionately on the elderly and on those with already compromised health.

The VLYL (Value of a Life Year Lost) approach takes the effects of age into consideration by dividing VSL by average expected years of life remaining (either discounted or not) to obtain WTP per year of life. A detailed discussion of the VLYL approach will be given in /European Commission 1997/. VLYL can be estimated from the VSL using equation 9-1:

$$VSL_a = VLYL(r) \cdot \sum_{i=a}^{T} \frac{{}_aP_i}{(1+r)^{i-a}} \qquad (9\text{-}1)$$

where

| | |
|---|---|
| $VSL_a$ | value of a statistical life |
| a | age of person whose VSL is being estimated |
| VLYL(r) | value of a life year lost (depending on discount rate r) |
| r | discount rate (e.g. 3% means r=0.03) |
| T | maximum life expectancy (100 years in our calculations) |
| ${}_aP_i$ | (conditional) probability that a person of age a will reach age i |

We propose to use the VLYL approach outlined above to value the risk of dying from the *acute illnesses*, and assume independence of age since this has not been tested sufficiently and there is no conclusive evidence that this is incorrect. Using the estimates of survival probabilities for the EU population /Markandya 1997/ in equation 9-1, and a VSL of 3.1 million ECU gives the results reported in table 7.2.

The main share of fatalities quantified by the exposure-response function based on the /Pope et al. 1995/ - study (which has Years of Life Lost (YOLL) as endpoint) will not occur immediately after exposure, but with a latency period, i.e. **chronic mortality**. To account for this, a "weighted" VLYL was calculated, assuming a uniform distribution of the impact over 30 years, i.e. from year 1 after exposure to year 30 the risk is increased by 1/30th of the value given by the exposure-response function. The weighted VLYL, which is dependent on the discount rate r, can be calculated from equation 9-2.

$$weighted\ VLYL(r) = \sum_{i=1}^{70} \frac{YOLL_i}{YOLL_{tot}} * \frac{VLYL(r)}{(1+r)^{i-1}} \qquad (9\text{-}2)$$

where

| | |
|---|---|
| $YOLL_i$ | aggregated years of life lost (YOLL) in year i |
| $YOLL_{tot}$ | total YOLL as: sum from year i=1 to 70 |
| VLYL(r) | value of a life year lost: depends on discount rate r, see table 8.1 |

Table 8.1 reports the results from calculating weighted VLYL. These values are the same as those used for other parts of the ExternE project, and also agree with the approach taken in ECs Green Accounting Study (GARP).

Similar life table calculations have been carried out for fatal cancer /Pilkington and Hurley 1997/ . In contrast to mortality from particles, where the exposure-response function give the change in mortality of the exposed population, the impact of exposure on survival of a population is not known. The survival probabilities for each age were scaled down by a factor $\lambda$, for each year after the latency period, to make the life expectancy changes equal to the estimated years of life lost. Then the value of a fatal cancer case (VC) can be calculated by equation 9-3:

$$VC = \sum_{i=1}^{70} YOLL_i \frac{VLYL(r)}{(1+r)^{i-1}} \qquad (9\text{-}3)$$

where

| | |
|---|---|
| l | average latency time |
| $YOLL_i$ | aggregated YOLL in year i (note that $YOLL_i = 0$ for i = 1,...,l) |
| VLYL(r) | value of a life year lost (depending on the discount rate, see table 8.1) |
| r | discount rate |

Since the pain and suffering of the individual in the period of illness prior to death is not accounted for, a cost of illness (COI) of 250,000 ECU is added to the VC estimate. The resulting values for different types of cancer is shown in table 8.1.

For *mortality in accidents*, where the age at death is on average 30-40 years we can use VSL, according to /Markandya 1997/. The VLYL is not appropriate since life years lost are large, and the studies from which VSL was estimated, deal with situations similar to the accidents that are being valued.

**Table 8.1:** Economic estimates for mortality impacts. 1995-prices.

| Impact | Value (ECU) | Comments |
|---|---|---|
| Mortality from particulates | | |
| - "Acute" YOLL | 98,000 | Discount rate = 0 % |
| | 155,000 | Discount rate = 3 % |
| | 312,000 | Discount rate = 10 % |
| - "Chronic" YOLL | 98,000 | Discount rate = 0 % |
| | 84,330 | Discount rate = 3 % |
| | 60,340 | Discount rate = 10 % |
| Mortality from cancer | | |
| - Leukaemia | 2,410,000 | Discount rate = 0 % |
| | 2,060,000 | Discount rate = 3 % |
| | 1,430,000 | Discount rate = 10 % |
| - Lung cancer | 1,820,000 | Discount rate = 0 % |
| | 1,330,000 | Discount rate = 3 % |
| | 670,000 | Discount rate = 10 % |
| - Stomach cancer | 1,720,000 | Discount rate = 0 % |
| | 1,240,000 | Discount rate = 3 % |
| | 620,000 | Discount rate = 10 % |
| - Nasal cancer | 1,620,000 | Discount rate = 0 % |
| | 1,100,000 | Discount rate = 3 % |
| | 500,000 | Discount rate = 10 % |
| Mortality in accidents | 3,100,000 | Value of a Statistical Life |

YOLL = Years of Life Lost

### 8.3.1 Morbidity

Most of the research on morbidity stems from US studies done back in the mid 1980s, see /Cropper and Freeman 1991/, chapter 6.4.2 for a good review and evaluation of these studies. We have reviewed three new European studies to see how they compare with the values currently used in ExternE for acute and chronic respiratory symptoms, which have been transferred from the US literature. Since many of the air pollutants specific to emissions from transportation are carcinogenic, the value for nonfatal cancer has been reconsidered. The recommended values are shown in table 8.2.

**Table 8.2:** Economic estimates for morbidity impacts. 1995-prices.

| Impact | Value (ECU) | Comments |
|---|---|---|
| Asthma | 105,000 | Same as chronic bronchitis |
| Bronchodilator usage | 37 | Same as a day of asthma |
| Chronic bronchitis | 105,000 | From this literature survey |
| Acute bronchitis | 225 | From this literature survey |
| Cerebrovascular HA | 7,870 | Value for general HA |
| Chronic cough | 225 | Same as acute bronchitis |
| Congestive heart failure | 7,870 | Value for general HA |
| Cough | 7 | From this literature survey |
| ERV for asthma | 223 | Value for general ERV |
| ERV for COPD | 223 | Value for general ERV |
| Hospital visits child croup | 223 | Value for general ERV |
| Chronic bronchitis HA | 7,870 | Value for general HA |
| Ischaemic heart disease | 7,870 | Value for general HA |
| Lower respiratory symptoms | 7.5 | Same as symptom day |
| Malignant neoplasm | | Same as lung cancer |
| | 1,820,000 | Discount rate = 0 % |
| | 1,330,000 | Discount rate = 3 % |
| | 670,000 | Discount rate = 10 % |
| Nonfatal cancer | 450.000 | COI x 1.5 |
| Respiratory HA | 7,870 | Value for general HA |
| Symptom Day | 7.5 | |
| Restricted Activity Days (RAD) | 75 | |
| Neoplasm, nonfatal | 450,000 | Same as non-fatal cancer |
| IQ in children | - | No data |

ERV= Emergency Room Visit, HA = Hospital Admission, COI= Cost of Illness, COPD = Chronic Obstructive Pulmonary Disease

### 8.3.2 Soiling

As a first approximation we suggest using the procedure suggested by /Pons et al. 1995/. They show that for the typical situation where damages are repaired by cleaning, which is the case of soot particulates from combustion, the amenity loss is equal to the cleaning costs. Thus, the total damage from soiling is twice the cleaning costs, as shown in equation 9-5. The annual soiling damage can then be calculated using equation 9-4.

$$S_i = a * P_i * \Delta TSP_i \qquad (9\text{-}4)$$

$$a = b * 2 \qquad (9\text{-}5)$$

where

$S_i$ = annual soiling damage at receptor location i

$P_i$ = number of persons in location i

| | | |
|---|---|---|
| $\Delta TSP_i$ | = | change in annual average TSP in $\mu g/m^3$ |
| a | = | WTP per person per year to avoid the soiling damage of a concentration of 1 $\mu g/m^3$ of particles. |
| b | = | Cleaning costs per person per year from a concentration of 1 $\mu g/m^3$ of particles: 0.5 ECU suggested by /Pons et al. 1995/ based on cleaning costs data from Paris |

### 8.3.3 Visibility

Following /Owe et al. 1995/, who reviewed the US valuation literature on visibility, we suggest using equations 9-6 and 9-7 for estimating the visibility damage costs. The implied annual average WTP per household for e.g. 5 and 10 % change in visual range (VR) can then be calculated at 6 and 12 ECU, respectively (with low and high estimates of 4-7.5 and 8.5-15).

$$HHWTP/year = b * \ln (VR2_i/VR1_i) \quad (9\text{-}6)$$

$$TVISD_i/year = b * \ln (VR2_i/VR1_i) * HH_i \quad (9\text{-}7)$$

where

| | | |
|---|---|---|
| HHWTP/year | = | annual WTP per household in area i for visibility changes in the year |
| $TVISD_i$/year | = | total annual WTP for visibility change in area i for residents of area i |
| $VR1_i$ | = | the starting annual average visual range (in km) |
| $VR2_i$ | = | the annual average visual range after the change in emission (measured in km) |
| b | = | estimated coefficient, a best central estimate of 150 was suggested by /Rowe et al. 1995/, with a range from 110 to 200, for the eastern US |
| $HH_i$ | = | number of households in the affected area i |

### 8.3.4 Other impacts

Regarding *acidification damages to ecosystems*, /Navrud 1997b/ reports the results of a Norwegian CV study that seeks to link the concepts of critical loads for fish populations and WTP. Based on a set of strict assumptions he concludes that the annual WTP per household to get increased fish stocks due to the current reduction plans for sulphur (S) is 0.64 and 2.45 mECU/t of "total" S and "critical load exceedance" S deposited, respectively.

From a review of the existing CV studies of *oil spills*, /Navrud 1996/ concludes that the best estimates seems to be an average WTP per household of 40, 75 and 125 ECU to avoid a small (< 1,000 t), medium (about 1,000 t) and large (>> 1,000 t) oil spill respectively. These estimates have to be multiplied by the number of affected households (including both users and non-users of the ocean and coastal areas) and converted from Net Present Values to annual

damage costs (since WTP was stated as a one-time amount). Then these costs must be multiplied by accident risks for these three groups of oil spills to come up with an annual expected damage value from oil spills. One should look at the possibility of converting this into a unit, e.g. costs of oil spills per vehicle-km, that could be used universally.

New empirical studies on the economic values for the low priority "non-energy" impacts of noise, road barrier effects and road accidents have also been reviewed. For road traffic noise new hedonic price studies indicate that the values for road traffic noise we now use might be too high.

# 9. SUSTAINABILITY INDICATORS

## 9.1 Introduction

The ExternE project has successfully used the damage function approach (DFA) and the willingness-to-pay (WTP) principle to value impacts from fuel cycle emissions, and now transport. However, it has been argued from the viewpoint of sustainable development that impacts on critical natural assets should be treated separately since there are no substitute for the life supporting functions they provide. These limitations to economic valuation has been recognised within the ExternE project, and in order to treat very long term, uncertain and potentially irreversible impacts like global warming, ecosystem damage and damage to historical buildings in a more satisfactory way, different sustainability approaches have been analysed. There are two distinct ways of introducing the sustainability concept in our damage cost study; either as exogenous variable in the form of e.g. physical sustainability indicators, or as an endogenous variable by trying to improve economic impact assessment and integrate the sustainability concept in the economic valuation estimate. These two approaches are often termed strong and weak sustainability, respectively. The DFA currently used in ExternE can be classified as a weak sustainability approach.

## 9.2 Towards an operational definition of sustainability

Even if there is no "textbook" definition of sustainability that commands widespread agreement, two issues are central; intergenerational equity and resource substitutability.

Given potential non-substitutabilities between services derived from natural capital $C_N$ (i.e. the total repository of service capacities from the natural environment), and man-made capital $C_M$ (i.e. manufactured capital, cultural and human capital), a complete sustainability criterion must address the flow of natural capital services as well as intergenerational fairness. Concepts of defining and measuring sustainable development can broadly be placed in four categories, ranging from very weak to very strong sustainability.

i) Very Weak Sustainability (VWS)

This sustainability rule merely requires that the overall stock of capital assets should remain constant over time. The rule is, however, consistent with anyone asset being reduced as long as another capital asset is increased to compensate. This concept is based on mainstream neo-classical economic theory and assumes that $C_M$ and $C_N$ are close substitutes.

(ii) Weak Sustainability (WS)

The "London School" has modified the very weak sustainability approach by introducing an upper bound on the assimilate capacity assumption, as well as a lower bound on the level of natural capital stocks that can support sustainable development, into the analysis (/Barbier and Markandya 1990/, /Pearce and Turner 1990/). The concept of critical natural capital (e.g. keystone species and keystone processes) has also been introduced to account for non-substitutability of certain types of natural capital (e.g. environmental support services) and man-made capital.

Figure 9.1illustrates the WS-approach. The vertical line depicts the critical level of natural capital (K). Substitution between $C_N$ and $C_M$ is possible only to the right of this line. To get increased economic growth (i.e. increased $C_M$), $C_N$ must be reduced. The VWS approach, on the other hand, assumes there is no such critical level, and that substitution is possible over the whole range of $C_N$ and $C_M$.

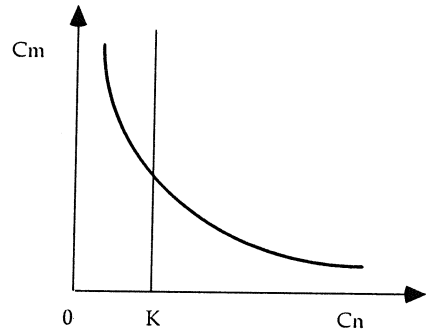



**Figure 9.1:** The Weak Sustainability (WS) view. Substitution between natural capital ($C_N$) and man-made capital ($C_M$) is possible only to the right of K, which is the critical level of $C_N$.

(iii) Strong Sustainability (SS)

This sustainability view, which is also termed the Ecological Economics Approach, requires that natural capital be constant, and the rule would be measured via physical indicators.

While akin to Safe Minimum Standards (SMS), SS is not the same. SMS says conserve unless the benefits foregone (social opportunity costs) are very large. SS says, whatever the benefits

foregone, some "critical", natural capital losses are unacceptable. SS need not imply a steady-state, stationary economy, but rather changing economic resource allocations over time which are not sufficient to affect the overall ecosystem parameters significantly. A certain degree of economic growth should therefore be possible via technical change, substitution of intermediate natural capital for natural capital, and environmental restoration investment.

(iv) Very Strong Sustainability (VSS)

This criteria is also called the Stationary State Sustainability, and calls for a steady-state economic system based on thermodynamic limits and the constraints they impose on the overall scale of the macroeconomy. Zero economic growth and zero population growth are required for a zero increase in the "scale" of the macroeconomy /Daly 1991/.

In practice, these various sustainability paradigms are less clearly defined and overlapping. However, to simplify, one could say that the weak sustainability criteria builds on monetary indicators (i.e. economic valuation of environmental impacts), while the strong criteria are based on physical indicators.

/Rennings and Wiggering 1995/ discuss and evaluate several weak and strong sustainability indicators. They conclude with a pledge for strong sustainability indicators, especially the methodological well-founded indicators of critical load and critical levels. However, they emphasise the need to take into consideration the costs and benefits of avoiding critical impacts. Therefore they propose a combination of weak and strong sustainability indicators (see also /Rennings and Hohmeyer 1997/).

## 9.3 Life Cycle Analysis (LCA) methods as strong sustainability indicators

We have evaluated the Swiss LCA (life cycle analysis) method, Ecological Scarcity (ECO), which can be based on the strong sustainability indicator of critical loads. We will compare ECO with two other LCA methods; the Dutch Environmental theme method (ET) and the Swedish Environmental Priority Strategies in Product Design Method (EPS) will also be described and compared to ECO.

The goal of all three methods is to set a one-dimensional value on resource use and emissions in order to calculate the total environmental impact of a product. In our case this product would be a transport task. The methods obtain their one-dimensional index in different ways, but the general relation between environmental loads, indices and the total impact of the product is given by:

$$TI(method) = \sum_j \left( LI_j(method) \cdot Load_j \right) \qquad \text{(10-1)}$$

where
TI (method) = total impact as calculated according to one of the three methods

$LI_j$ (method) = load index j according to one of the three method i.e. environmental impact per mass unit of emission j
$Load_j$ = environmental load of j (emission or resource use) of the product, mass unit
j = substance
method = ECO, ET or EPS.

Since the ECO method seems to be the method that most explicitly calculates the impacts this is the LCA method closest to the Damage Function Approach used in ExternE. However, ECO takes an oversimplified approach to dispersion modelling. Also, instead of monetary valuation the ECO method uses the strong sustainability indicator of ecologically critical loads (or political goals for substances where critical loads are not defined) is used to value impacts.

Rather than using the ECO method in ExternE Transport, we have looked at the possibility of developing a method that combines the strong sustainability indicator of critical loads and the weak sustainability indicator of economic damage estimation in a more direct and transparent way.

## 9.4 An attempt to link critical loads and economic valuation

Critical loads for sulphur (S) and Nitrogen (N) with regards to acidification and the impacts on fish populations were used in a Contingent Valuation (CV) study to illustrate the possibility of linking critical loads with economic valuation, and thus integrate the strong sustainability dimension into the Damage Function (DF) approach adopted in ExternE.

/Navrud 1997/ reports the results from a nation-wide Contingent Valuation (CV) survey with in-person interviews of 1,000 Norwegian households. The respondents were asked about their willingness-to-pay (WTP) for a specified increase in areas with no damages to fish populations. Two colour maps depicting the situation in 1990 and 2010 were shown to illustrate the effects on Norwegian fish populations from the current international agreement on reductions in sulphur dioxide emissions, which is based on the critical load concepts. (No change in the Nitrogen oxides emissions was assumed). The colour maps were based on calculations by the Norwegian Institute of Water Research (NIVA) of critical loads of acidity and critical load exceedance for sulphur and nitrogen, and the predicted reduction of current international agreement on sulphur reductions. Then estimated dose-response functions for three damage classes of the fish population (1) Good/unchanged fish population, (2) reduced fish population, and (3) Lost fish population were used to calculate the reduced fish damage.

If we assume a linear relationships between the reduced deposition and the economic value, we can calculate the economic value per tonne of reduced deposition of S: 1,276 ECU per ton of total S, and 4,850 ECU per ton of "critical load exceedance" S. The corresponding numbers on a per household basis is 0.64 and 2.45 mECU per ton of "total" S and "critical load exceedance" S, respectively. The economic value on a per "critical load exceedance" S - basis

is the preferable measure if we want to transfer these estimates to other locations in Europe. However, the uncertainty is such benefit transfers are large.

## 9.5 Sustainability indicators for global warming, ecosystems and historical building

For acidification effects on ecosystems /Hornung 1996/ suggests using the change in the area over which critical loads for the pollutant in question is exceeded as a result of the transport task. Impacts on forests and surface water can be calculated using the UN-ECE critical load maps (databases) for acidity and nitrogen for forests and surface waters, and presented as values for EMEP grid cells. However, as /Mayerhofer 1997/ points out, the strong sustainability indicator of "exceedance area difference" will not be a meaningful indicator for facility analysis (and transport tasks), because the increases in deposition levels are so small that the exceedance area difference in most cases will be zero.

/Mayerhofer 1997/ also discusses the use of critical loads and levels as a sustainability indicator for corrosion of materials. Contrary to acidification and eutrophication effects on ecosystems, a no-effect level does not exist for the impacts of airborne pollutants because there is no self-restoring process as for ecosystems. Mayerhofer recommends not to use an "acceptable level" indicator to evaluate the loss of authenticity from material loss of historical monuments.

In the ExternE Core project /Rennings and Hohmeyer 1997/ reviewed the literature on strong and weak sustainability indicators for global warming impacts. They conclude that the outcomes of studies estimating economic and social impacts in monetary units are very uncertain, and that the uncertainty do not disappear when alternative approaches of strong sustainability norms for "tolerable stress levels" are used.

They also find that even the strong sustainability indicators in terms of normative target approaches often depend on monetary values for defining tolerable stress levels. On the other hand a complete substitution of normative judgement by damage cost valuation techniques will hardly be possible since these techniques themselves contain several normative assumption (of welfare theory). They recommend to use both the strong and weak sustainability concepts as part of broader approaches of integrated assessment models, and mention multi-critieria analysis, ecological economics, and a broader interpretations of cost benefit analyses as ways of achieving this.

## 9.6 Conclusion

We have reviewed the literature on the definition of sustainable development and the different forms of strong and weak sustainability. Based on a discussion of several weak and strong sustainability indicators, we recommend a combination of the two in ExternE. We also discuss life cycle analysis (LCA) indicators, but find them unsuitable as sustainability indicators in ExternE. To show that it is possible to operationalize a combination of weak and strong

sustainability indicators we have carried out an empirical study on acidification effects on freshwater fish in Norway, where we combine the strong sustainability indicator of critical loads into the weak sustainability approach of economic valuation. We also review the results from the ExternE "Core" project on sustainability indicators for global warming, ecosystems and historical buildings. Our recommendations are consistent with the ExternE Core Project results.

# 10. CASE STUDIES FRANCE

## 10.1 Introduction

Using the methodology of ExternE Transport, we have evaluated the costs of air pollution damage from transport by way of two case studies in France: a trip across Paris and a trip from Paris to Lyon. The transport vehicles include a conventional petrol car, a petrol car with three-way catalyst, a diesel car, a diesel bus, and the TGV train (train grande vitesse = French high speed train). The air pollution damages are compared with the price of fuel and with the number of years of life lost due to traffic accidents.

## 10.2 Emissions

The diesel and petrol vehicles considered here are representative of the fleet of automobiles currently operating in France. In fact, to be more general, we have used for each type of vehicle (be it diesel or petrol) average characteristics, which are based on the different car models existing in France in 1995. Average parameters include, for example, engine size and power, vehicle age and weight, odometer reading and airborne pollutant emission rates. The emissions are calculated for the actual driving conditions, using detailed data for the appropriate driving cycles. The total emissions for each trip are shown in table 10.1.

**Table 10.1:** Vehicle emissions.

a) for 43.3 km trip from Charles-de-Gaulle to Orly airport expressed in g per passenger, assuming occupancy: 1 person/car:: 25 persons/bus.

| | CO | VOC | NOx | $SO_2$ | $CO_2$eq | Fine Particles |
|---|---|---|---|---|---|---|
| Petrol car 3way cat. | 122.8 | 11.08 | 29.5 | 1.65 | 11233 | 0.75 (*) |
| Diesel car | 40.6 | 7.25 | 32.4 | 1.45 | 9279 | 7.52 |
| Diesel bus | 22.1 | 17.75 | 55.0 | -- | 3603 | 4.76 |

b) for 481 km trip from Paris to Lyon expressed in g per passenger, assuming occupancy: 1 person/car;: 25 persons/bus.

| | CO | VOC | NOx | $SO_2$ | $CO_2$eq | BaP ($x10^6$) | Fine Particles |
|---|---|---|---|---|---|---|---|

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Petrol car no cat. | 6341 | 780 | 1617 | 14.2 | 81813 | 471 | 14.4 (*) |
| Petrol car 3way cat. | 622 | 59 | 379 | 14.5 | 97383 | 19.7 | 7.2 (*) |
| Diesel car | 268 | 54 | 299 | 13.5 | 84744 | 155 | 72.1 |
| Diesel bus | 245.4 | 197.3 | 611.0 | | 40028 | | 52.9 |

(*) Estimated from data by /Heywood 1988/.

To quantify the energy related external costs of the TGV, we have assumed an electricity use of 50 kWh per kilometre with an average external cost of 0.0022 ECU per kWh of electricity. This is based on an energy mix in France of 4% coal (impact of 23.9 mECU/kWh), 4% oil (impact of 29 mECU/kWh), 12% hydro and 80% nuclear (impact of 0.098 mECU/kWh at a 3% discount rate /Rabl et al. 96/). It is important to note that damage costs are highly sensitive to the energy mix used to produce the electricity and, therefore, can vary significantly amongst different countries. Hence, the damage cost per trainload is equal to 110 mECU/vkm, and assuming an average capacity of 500 people for the Paris to Lyon journey, the damage cost per person comes out to 0.22 mECU/pkm.

## 10.3 Impact Evaluation

Regional scale impacts to crops, building surfaces and health due to aerosols (sulphates and nitrates) are calculated internally by ECOSENSE. For the remaining pollutants, the local concentrations are calculated by ROADPOL and the impacts by PATHWAYS. ECOSENSE is used for evaluating the regional scale atmospheric transport of pollutants and their resulting damages.

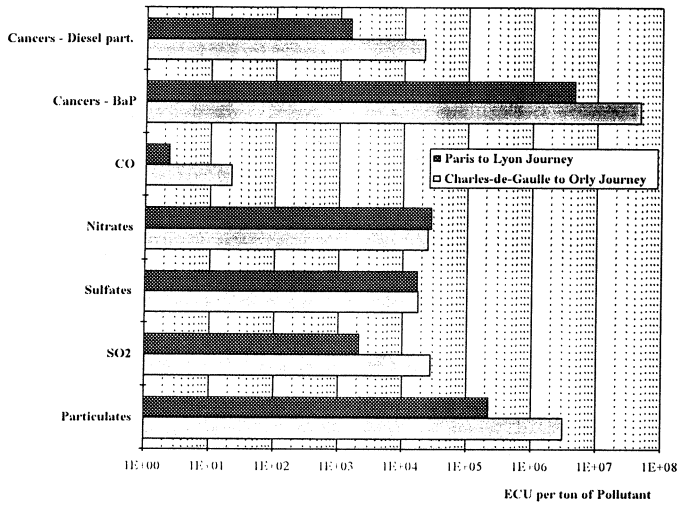



**Figure 10.1:** Comparison of damage costs per ton of pollutant for Paris-to-Lyon (mostly rural) and Charles de Gaulle-to-Orly airports (urban).

We also quantify damage costs for ozone and global warming. These impacts are calculated on a regional or global scale only. With respect to greenhouse gases, we use the value of 15.5 ECU per ton of equivalent $CO_2$ ($CH_4$ and $N_2O$ have a global warming potential of 21 and 310, respectively). The impact for global warming is based on the IPCC 1996 study and represents the arithmetic average for the IPCC proposed range. For ozone, we use the damage costs of 1500, 930 and 130 ECU per ton of precursor NOx, VOC and $CH_4$, respectively, as determined by /Rabl and Eyre 1997/. These figures include impacts on crops (21%) and health (47% morbidity and 32% mortality).

The results, expressed as ECU/t of pollutant, are shown in figure 10.1. The comparison of the urban (Charles de Gaulle-to-Orly) and the inter-city (Paris-to-Lyon) trips shows that site dependence is strong for the primary air pollutants: the damage from particulates is an order of magnitude larger for the former. For nitrate and sulphate aerosols the variation with emission site is small. For ozone there is some site dependence but we have no data; the numbers of /Rabl and Eyre 1997/ are European averages based on EMEP.

When the cost per tonne is combined with the emissions data one obtains the damage cost per trip (note that the result shown here are obtained with detailed emissions data for each section and driving cycle, rather than averaged data). The results are shown in figure 10.2 for *Charles de Gaulle-to-Orly* airports and in figure 10.3 and figure 10.4 for the *Paris-to-Lyon* trip,

expressed as mECU per passenger km assuming occupancy numbers as indicated. To highlight the great uncertainties, figure 10.5 suggests typical error bars.

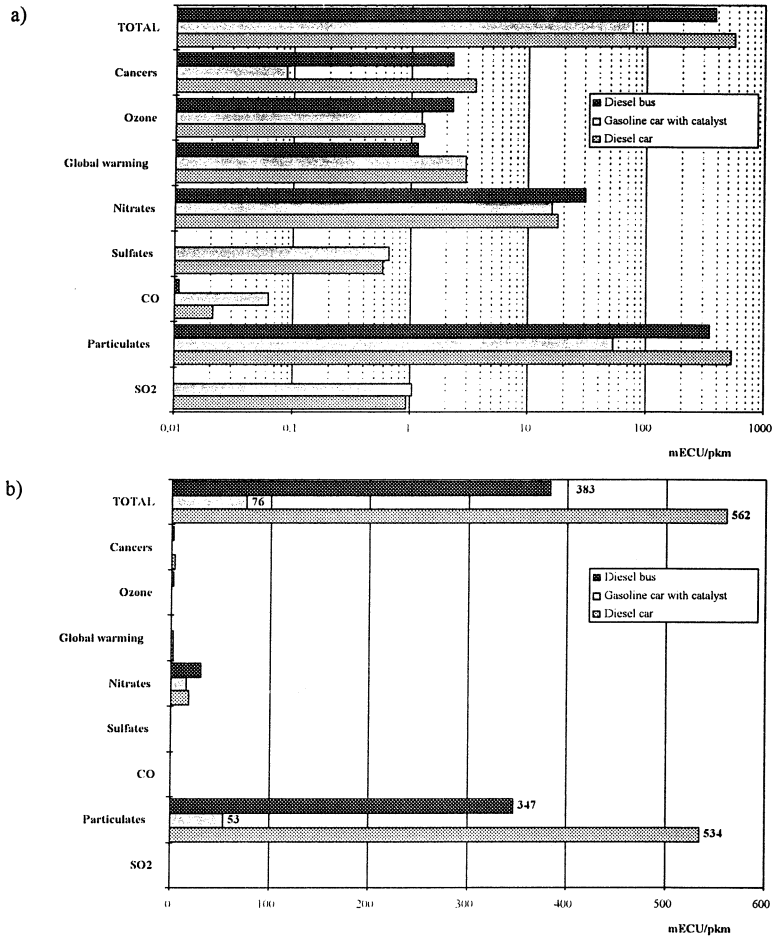



**Figure 10.2:** External costs for the journey from *Charles-de-Gaulle to Orly* airports (Paris, France) using various transport vehicles. Local impact (greater Paris area) accounts for 95% of the total damage. Assuming occupancy: 1 person/car, 25 persons/bus. a) logarithmic scale, b) linear scale.

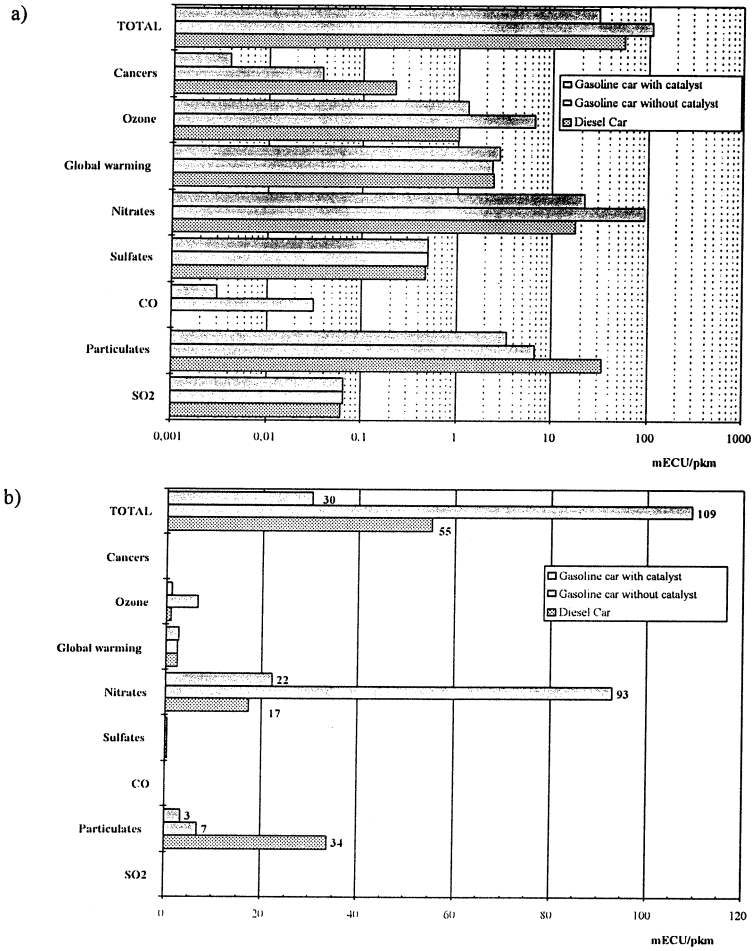
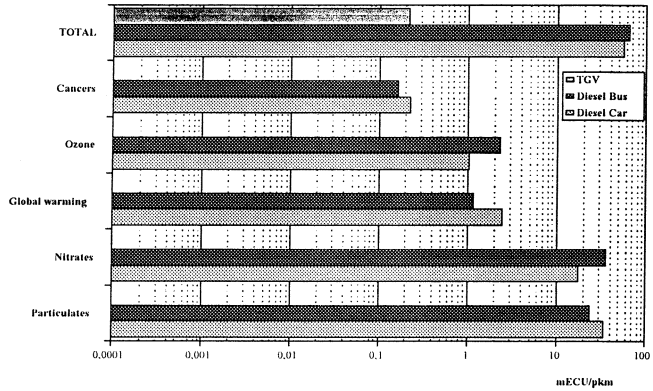



**Figure 10.3:** Transport costs for moving a *single* passenger on a one-way trip from *Paris to Lyon* using different transport vehicles. a) logarithmic scale, b) linear scale.

a)




b)

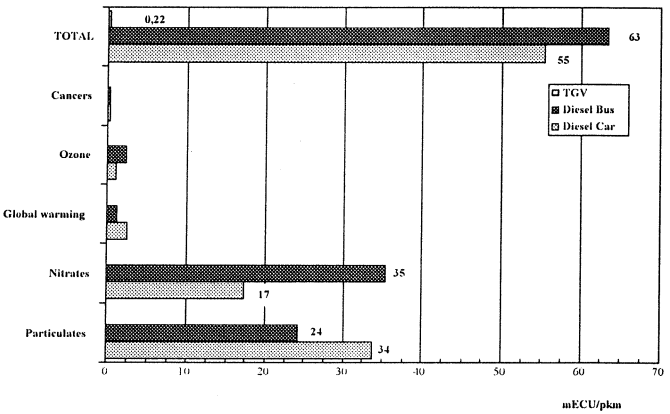



**Figure 10.4:** Damage costs per passenger: diesel car vs diesel bus and TGV train (*Paris-Lyon trip*). Electricity production in France: 80% nuclear, 12% hydro, 4% each coal and oil. Assuming occupancy: 1 person/car; 25 persons/bus, 500 persons/train. a) logarithmic scale, b) linear scale.

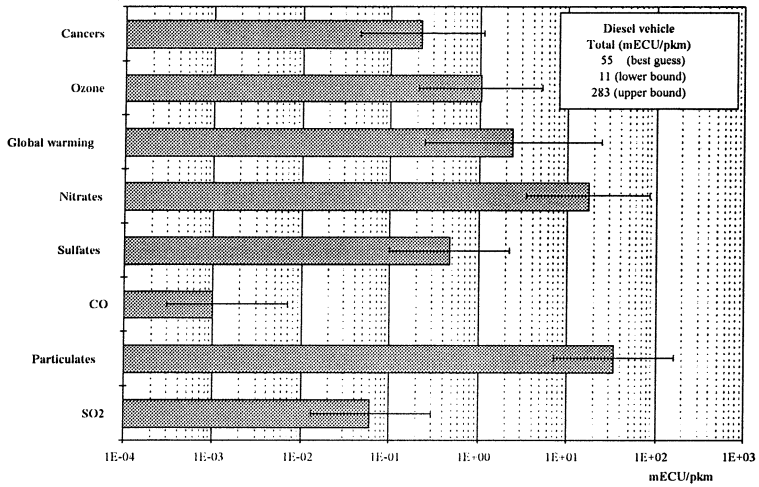



**Figure 10.5:** Typical uncertainty ranges for impacts for the Paris-Lyon trip. (Note: Nitrate costs could be zero, uncertainty label B?).

## 10.4 Conclusions

The magnitude of the external costs due to air pollution from motor vehicles can be put in perspective by making several comparisons, in instance with the price of fuel. Another natural comparison is the number of YOLL due to air pollution from vehicles with the number of YOLL due to traffic accidents in France. By any of these comparisons the external costs due to air pollution from motor vehicles are significant.

It is also instructive to compare the costs due to different pollutants. Carcinogens, despite their high specific toxicity, are negligible compared to the classical air pollutants because the emitted quantities are very small. Costs due to particulates and NOx appear to be more important than global warming. Unfortunately the largest item, nitrate aerosols from NOx emissions, also has the greatest uncertainty because it is based on inference rather than direct epidemiological evidence.

# 11. CASE STUDIES GERMANY

Following the impact pathway methodology, energy-related external costs of road and rail transport were estimated within two case studies. Case study 1 is a regional transport task from Stuttgart to Mannheim, two major industrial centres in the south-western part of Germany, leading mainly through extra-urban areas. Passenger and goods transport via motorway and railway is assessed. Case study 2 is located in an urban environment only: transport from a suburb of Stuttgart to the very city centre. Public transport on this route is well developed and the assessment comprises bus, suburban railway and tram lines besides individual road transport.

## 11.1 Road transport

**Table 11.1:** Road transport vehicle technologies considered.

| Vehicle Type | Fuel | Characterisation | Emission Standard | Identifier |
|---|---|---|---|---|
| Car | Petrol | no catalyst | ECE 15/04 | C-80ies no cat. |
| | Petrol | TWC - "old" | before 1987 | C-80ies TWC |
| | Petrol | TWC - "modern" | before EURO2 | C-90ies TWC |
| | Petrol | TWC - "new technology" | EURO2 | C-EURO2 |
| | Diesel | "old" | years 1986 to 1988 | C-80ies Diesel |
| | Diesel | "modern" | App. XXIII StVZO[1)] | C-90ies Diesel |
| Van | Petrol | no catalyst | after 1981 | V-80ies no cat. |
| | Petrol | TWC | | V-TWC |
| | Petrol | TWC - "new technology" | EURO2 | V-EURO2 |
| | Diesel | "modern" | App. XXIII StVZO[1)] | V-90ies Diesel |
| | Diesel | "new technology" | EURO1 | V-EURO1 Diesel |
| Lorry | Diesel | GVW <= 7.5 t - "old" | 1980ies | L-80ies<7.5 |
| | Diesel | GVW <= 7.5 t - "modern" | EURO1 | L-EURO1<7.5 |
| | Diesel | GVW 14 to 20 t - "old" | 1980ies | L-80ies 14-20 |
| | Diesel | GVW 14 to 20 t - "mod." | EURO1 | L-EURO1 14-20 |
| Lorry + Trailer | Diesel | GVW 20 to 28 t "old" | 1980ies | L-80ies Trailer |
| Articulated Train | Diesel | GVW > 32 t "old" | 1980ies | Articulated Train |
| Coach | Diesel | GVW > 16 t - "old" | 1980ies | Coach 80ies |
| | Diesel | GVW > 16 t - "modern" | EURO1 | Coach EURO1 |
| Bus | Diesel | GVW < 20 t - "old" | 1980ies | Bus 80ies |
| | Diesel | GVW < 20 t - "modern" | EURO1 | Bus EURO1 |

[1] German emission standard for new diesel vehicles until the end of 1996 (Appendix XXIII StVZO)

TWC = Three way catalyst; GVW = Gross vehicle weight

A broad range of vehicle technologies was included to allow analysis of the effects of technological progress, whereby the focus was put on the existing fleet. The technologies, which are listed in table 11.1 are characterised by the emission standard they comply with.

Vehicle emissions are mainly determined by road type, road gradient and traffic situation (e.g. congested, free flow). For emission calculation, emission factors provided by the German Federal Environmental Office (Umweltbundesamt) /Infras 1995/ were used, which take into account these parameters. In addition to warm engine emissions, for cars and vans cold start emissions were considered where available.

Emission factors for $SO_2$, Benzene, $CH_4$, Formaldehyde, and 1,3-Butadiene were derived from fuel use or VOC emissions using split factors.

**Table 11.2:** Average emission factors in g/km (including cold start emissions) for selected vehicle technologies for the inter-city case study (Stuttgart - Mannheim)

| Technology | PM | NOx | $SO_2$ | CO | VOC | Benzene | BaP | 1-3 Butadiene | $CO_2$ |
|---|---|---|---|---|---|---|---|---|---|
| C-EURO2 | 0.005 | 0.24 | 0.013 | 0.609 | 0.044 | 0.0037 | 2.6E-07 | 1.8E-04 | 153.4 |
| C 90ies TWC | 0.005 | 0.399 | 0.013 | 1.549 | 0.107 | 0.009 | 2.6E-07 | 4.3E-04 | 153.4 |
| C 80ies no cat. | 0.005 | 2.31 | 0.012 | 10.2 | 1.17 | 0.064 | 2.6E-07 | 7.2E-03 | 150.2 |
| C-90ies Diesel | 0.055 | 0.478 | 0.117 | 0.293 | 0.049 | 0.00093 | 6.7E-07 | 5.4E-04 | 120.3 |
| C-80ies Diesel | 0.078 | 0.479 | 0.125 | 0.359 | 0.072 | 0.0014 | 6.7E-07 | 7.9E-04 | 128.2 |

**Table 11.3:** Average emission factors in g/km (including cold start emissions) for selected vehicle technologies for the urban case study (Stuttgart)

| Technology | PM | NOx | $SO_2$ | CO | VOC | Benzene | BaP | 1-3 Butadiene | $CO_2$ |
|---|---|---|---|---|---|---|---|---|---|
| C-EURO2 | 0.009 | 0.22 | 0.018 | 2.65 | 0.252 | 0.021 | 2.3E-06 | 1.0E-03 | 191.6 |
| C 90ies TWC | 0.009 | 0.42 | 0.018 | 6.45 | 0.593 | 0.05 | 2.3E-06 | 2.4E-03 | 191.6 |
| C 80ies no cat. | 0.009 | 1.54 | 0.018 | 20.18 | 3.858 | 0.21 | 2.3E-06 | 2.3E-01 | 192.6 |
| C-90ies Diesel | 0.084 | 0.587 | 0.15 | 0.83 | 0.138 | 0.0026 | 1.4E-06 | 0.0015 | 137.8 |
| C-80ies Diesel | 0.123 | 0.583 | 0.158 | 0.786 | 0.167 | 0.0032 | 1.4E-06 | 0.0018 | 147.3 |

## 11.2 Rail Transport

The railway line from Stuttgart to Mannheim is one of the busiest lines of the German railway network and is electrified. Furthermore, this line is especially constructed for high speed trains. Hence, the case studies only consider electric traction for both passenger and goods trains.

**Table 11.4:** Specification of the train types (Source: /Maibach et al. 1995/)

| Train type | Composition | Maximum Load | Average Load | Energy Use |
|---|---|---|---|---|
| | | persons/tonnes | persons/tonnes | $MJ_{el}$/train km |
| Short-Distance Train | Loc. + 3 coaches | 193 p | 32 p | 19.0 |
| Inter-city Train | Loc. + 10 coaches | 739 p | 229 p | 57.6 |
| High-Speed Train | 2 Loc. + 12 coaches | 771 p | 424 p | 83.2 |
| Tram | Railcar + coach | 160 p | 28 p | 17.5 |
| Goods Train | Loc. + 28 wagons | 746 t | 319 t | 111.0 |

Loc. = Locomotive

For the inter-city passenger transport task three train types were considered: short-distance train, inter-city train and high speed train. On this route a goods train was also assessed. For the urban transport task passenger transport by tram was examined. Table 11.4 shows the composition and characteristics of the train types.

## 11.3 Impact calculation

The calculation followed the impact pathway methodology with the stages emission modelling, dispersion modelling, estimation of physical impacts using exposure-response functions (ERF) and finally monetising of impacts.

Pollutant concentration increments in the environment due to vehicle emissions or power plant emissions were estimated using different Gaussian dispersion models (e.g. ROADPOL for vehicle emissions) on the local scale and the trajectory model WTM, which includes chemical transformations, on the regional scale.

ERFs for human health, crops and materials were applied using corresponding receptor inventories. Monetary values for the individual endpoints, based mainly on willingness-to-pay surveys, lead finally to the cost estimate. Important to mention is that valuation of mortality impacts are based on the *Years-of-Life-Lost* (YOLL) approach.

Results for rail transport are based on average values for electricity consumption of the different train types. External cost estimates for German power plants were used, according to the energy mix of the train types.

## 11.4 Results

Different air pollutants impose different burdens on the environment. Road vehicles and power plants release a mixture of pollutants, mainly into the air. Therefore the damages for the transport modes and the individual vehicle technologies vary in their composition.

Within passenger cars, non-catalyst, three way catalyst (TWC) and diesel cars can be distinguished. Damages from non-catalyst cars are dominated by nitrates (ca. 80%) which are attributed to their high $NO_x$ emissions. TWCs reduce the emissions of carbon monoxide,

hydrocarbons and nitrogen oxides: nitrate damages decrease to ca. 47% of the total. The share of global warming increases from about 5% (non-catalyst) to roughly 30%, while the total amount is reduced (see table 11.5. Damages from diesel cars have a different composition. More than half of the total is caused by particles. Nitrates have a share of about 30%.

**Table 11.5:** Damage costs (vehicle use only) for passenger transport modes, given as 'best estimate'. The uncertainties in the estimates is of class B (a factor of 4-6).

| Technologies / modes | inter-city (case study 1) | | | urban (case-study 2) | | |
|---|---|---|---|---|---|---|
| mECU | per vkm | per pkm | | per vkm | per pkm | |
| | | average | full load | | average | full load |
| Car | | | | | | |
| C-EURO2 | 7.60 | 5.21 | 1.9 | 10.02 | 6.86 | 2.51 |
| C-90ies TWC | 10.19 | 6.98 | 2.55 | 12.24 | 8.38 | 3.06 |
| C-80ies TWC | 17.63 | 12.08 | 4.41 | n.q. | n.q. | n.q. |
| C-80ies no cat. | 43.22 | 29.60 | 10.81 | 37.54 | 25.71 | 9.39 |
| C-90ies Diesel | 24.37 | 16.69 | 6.09 | 49.12 | 33.64 | 12.28 |
| C-80ies Diesel | 30.24 | 20.71 | 7.56 | 65.28 | 44.71 | 16.32 |
| Coach[1] | | | | | | |
| Coach 80ies | 369.93 | 9.25 | 7.40 | 912.60 | 22.82 | 18.25 |
| Coach EURO1 | 278.10 | 6.95 | 5.56 | 697.04 | 17.43 | 13.94 |
| Public service bus[1] | | | | | | |
| 80ies | n.q. | n.q. | n.q. | 633.92 | 41.54 | 7.23 |
| EURO1 | n.q. | n.q. | n.q. | 484.83 | 31.77 | 5.53 |
| Train[2] | | | | | | |
| Local train | 165.14 | 5.16 | 0.86 | — | — | — |
| Inter-city train | 500.62 | 2.19 | 0.68 | — | — | — |
| High speed train | 723.12 | 1.71 | 0.94 | — | — | — |
| Tram | — | — | — | 19.2 | 0.69 | 0.12 |

[1] diesel fuelled.
[2] electric traction, fuel production damages are included.

Because of the nature of the pollutants (primary and secondary pollutants) and the emission characteristics of road vehicles (emissions near the ground), the importance of the local and the regional environment differs. Damages from diesel vehicles ('particles') roughly double, comparing the inter-city to the Stuttgart urban drive, while the emissions are only increased on the urban drive by a factor of ca. 1.5-1.6. Based more on secondary and global pollutants, damages from petrol cars are less sensitive to the location of the drive.

For a comparison of passenger transport modes, the external costs per passenger kilometre has to be regarded. About 1.5 persons are transported in a car in the average, while public busses are only occupied by 17,4% in average in Germany (ca. 16 persons). For this reason, and

because of the high particle emissions of diesel buses, most passenger cars cause lower damages from operation than public service buses, per passenger. Trains cause lower damages per pkm than road vehicles. This holds especially for the high-speed train because of the high occupancy rates (see table 11.4). Tram has the lowest value of all modes assessed, mainly because of the high share of nuclear power generation.

**Table 11.6:** Damage costs (vehicle use only) for goods transport, given as 'best estimates'. The uncertainties in the estimates is of class B (a factor of 4-6).

| Technologies / modes | inter-city (case study 1) | | | urban (case-study 2) | | |
|---|---|---|---|---|---|---|
| mECU | per vkm | per tkm | | per vkm | per tkm | |
| load | | average | full | | average | full |
| Light goods vehicles | | | | | | |
| V-80ies no cat. | 53.90 | 179.68 | 53.90 | 39.42 | 131.39 | 39.42 |
| V-TWC | 20.04 | 66.79 | 20.04 | 19.20 | 64.01 | 19.20 |
| V-EURO2 | 9.93 | 33.11 | 9.93 | 11.90 | 39.66 | 11.90 |
| V-90ies Diesel | 61.01 | 203.37 | 61.01 | 114.71 | 382.37 | 114.71 |
| V-EURO1 Diesel | 42.03 | 140.09 | 42.03 | 81.95 | 273.18 | 81.95 |
| Heavy goods vehicles[1] | | | | | | |
| L-80ies <7.5 | 165.54 | 103.46 | 41.38 | 354.95 | 221.85 | 88.74 |
| L-EURO1 <7.5 | 127.38 | 79.61 | 31.85 | 279.06 | 174.41 | 69.76 |
| L-80ies 14-20 | 313.30 | 82.45 | 32.98 | 795.38 | 209.31 | 83.72 |
| L-EURO1 14-20 | 238.53 | 62.77 | 25.11 | 614.91 | 161.82 | 64.73 |
| L-Trailer 80ies 20-28 | 305.78 | 43.68 | 17.47 | 645.88 | 92.27 | 36.91 |
| Art. train 80ies >32 | 478.67 | 44.32 | 17.73 | 1041.64 | 96.45 | 38.58 |
| Goods train[2] | 964.74 | 3.02 | 1.29 | — | — | — |

[1] diesel fuelled.
[2] electric traction, fuel production damages are included.

Goods vehicles show basically the same damage pattern as cars, petrol/diesel fuelled light goods vehicles (LGV) and diesel fuelled heavy goods vehicles (HGV) respectively. The higher the cubic capacity, the higher are the emissions and of course the damages per vehicle kilometre. Looking at the load however, the higher the load capacity, the lower are the damage estimates per ton kilometre, considering European average load factors. Goods trains cause one order of magnitude lower damages than comparable road transport means. LGVs have in comparison an unfavourable high result. The estimates are given in table 11.6.

Externalities result not only from the vehicle use itself. Different stages of the life cycle of the vehicle, i.e. production, maintenance and disposal, impose burdens to the environment. Furthermore, the use of a transport means requires fuel production and the provision of infrastructure. Estimates are given in table 11.7, based on average emissions given by Maibach et al.

**Table 11.7:** Damage costs for life cycle stages of transport means.

| | fuel production | vehicle production | vehicle maintenance and disposal | infra-structure | total | total |
|---|---|---|---|---|---|---|
| mECU | /vkm | /vkm | /vkm | /vkm | /vkm | /pkm /tkm |
| **Road transport** | | | | | | |
| petrol car | 5.5 | 5.7 | 0.9 | 1.1 | 13.2 | 9.1[2] |
| diesel car | 2.7 | 5.7 | 0.9 | 1.1 | 10.4 | 7.1[2] |
| LGV[1] | 8.1 | 5.2 | 1.8 | 0.8 | 15.9 | 52.9[3] |
| Lorry 16t | 14.4 | 5.4 | 1.5 | 9.7 | 31.0 | 8.2[3] |
| Lorry 28t | 17.6 | 7.3 | 3.2 | 18.1 | 46.1 | 6.6[3] |
| Lorry 40t | 21.2 | 12.6 | 7.2 | 27.8 | 68.8 | 6.4[3] |
| Coach | 15.5 | 8.1 | 1.5 | 15.2 | 40.2 | 1.0[2] |
| Bus | 19.0 | 8.1 | 1.5 | 11.5 | 40.1 | 2.6[2] |
| **Rail transport** | | | | | | |
| Local train | | 11.8 | 3.2 | 161.1 | 176.1 | 5.5[2] |
| Inter city train | | 25.3 | 7.2 | 519.2 | 551.7 | 2.4[2] |
| High speed train | | 37.9 | 9.6 | 1449.4 | 1449.9 | 3.5[2] |
| Tram | | 7.2 | 0.9 | 75.1 | 83.2 | 3.0[2] |
| Goods train | | 91.7 | 51.4 | 1450.2 | 1593.3 | 5.0[3] |

[1] LGV = Light goods vehicle
[2] external costs per passenger kilometre based on average occupancy rates.
[3] external costs per ton kilometre based on average load factors.

It has to be noted, that the presented results give not the full external costs of transport. Only energy-related effects were regarded, above all caused by airborne pollutants and greenhouse gases. Other impacts, e.g. noise impacts and accidents have not been addressed in this project, and therefore are not included in the results.

# 12. CASE STUDIES GREECE

The transport sector in Greece has a share of 27.9% in the total energy consumption, the major part of which (about 85%) represents road transportation. The transport sector contributes greatly to the total emissions of the country, and because a large part of these emissions occur in metropolitan areas, this sector is the main contributor to the of urban atmospheric pollution.

The purpose of this case study is to apply the ExternE methodology for transport, and to estimate the damages caused by a variety of transport means and fuels used by the transport sector in Greece. The case studies were selected on the basis of transport tasks representing basic and common itineraries, which can be aggregated on a national level at a following stage. Five case studies were defined for implementation:

**Urban passenger transport by car**

A typical trip for people going to work in Athens was defined. Athens is a city where most of the financial and trading activities take place in the centre of the city. Due to the lack of an efficient public transport system, there is a wide use of cars. The transport task is to travel from a suburb to the centre of Athens, using an average, for the Greek case, passenger car of 1.4l capacity, fitted with a three way catalytic converter, consuming unleaded petrol. The total length of the trip is 8.85 km.

**Urban passenger transport by car** and **trolley bus**

Electric trolley buses are widely used by the public transport company in Athens. They are equipped with 110-150 kW electric motors. Their advantage is that they have zero direct emissions, but they are slow in moving and often cause traffic congestion. The itinerary of the transport task is the same with the one of the „urban passenger transport by car" case study, but is made up of two blocks. The first block of the task is to travel by car to the trolley bus station (4.87 km), and the second block is to travel the rest of the route (3.98 km) by trolley bus.

**Regional passenger transport by car**

Travelling between Athens and Thessaloniki is a typical trip for business purposes. Thessaloniki is the second major city of Greece located in the northern part of the country, about 500 km away from Athens. The trip takes place in a motorway and is accomplished by a passenger car of 1.4l capacity, fitted with a three way catalytic converter, consuming unleaded petrol. The total length of the trip is 492 km.

**Regional goods transport by truck**

The road from Athens to Thessaloniki is the most important trading route in Greece connecting the two major cities of Greece. In addition, this road is part of one of the alternative trading routes connecting Athens and the agricultural areas of Greece with the European market. The trip takes place in the motorway connecting the two cities and is accomplished by a diesel heavy duty truck. The total length of the trip is 492 km.

**Regional passenger transport by train**

The transport task chosen is to travel from Athens to Thessaloniki by train. The trains used in this line are equipped with diesel engines and have two railcars and two wagons with total passenger capacity of 296 people. The total length of the trip is 488 km.

Analytical data concerning traffic flow volume and speed, demographic and meteorological features of the areas crossed by the routes of the transport tasks were collected. The transport tasks were then disaggregated into building blocks (task/technology/location combinations) in order to pinpoint differences between them.

The impact assessment requires an estimation of the emissions and their dispersion in the environment. The calculation of the spatial distribution of the emissions was performed on a Geographic Information System (GIS). The use of GIS is recommended especially in urban areas, where due to the dense road networks and the traffic fluctuations the pollutant emissions appear significant spatial and temporal variations. Spatial databases of the roads and railways networks, containing annual average traffic flow and street co-ordinates data, were created on GIS environment. The pollutants emissions for each transport task were estimated using the CORINAIR methodology. The emission factors used for the estimation of the pollutant emissions are:

- Passenger cars
- Speed dependent emission factors for CO, VOC, $NO_X$
- Speed dependent fuel consumption
- Emission factors for $CH_4$, $N_2O$, $NH_3$, $SO_2$, Pb, $CO_2$
- Cold start emissions
- Evaporative emissions
- Trucks
- Driving mode (urban, rural, highway) dependent emission factors for CO, VOC, $NO_X$ and particulate matter
- Driving mode dependent fuel consumption
- Emission factors for $CH_4$, $N_2O$, $NH_3$, $SO_2$, Pb, $CO_2$

- Trains
- Speed dependent emission factors for CO, VOC, $NO_X$, $CH_4$, $N_2O$, $NH_3$, $SO_2$ and particulate matter

The damages of internal combustion engine vehicles were estimated at local, regional and global level. Local level damage assessment includes impacts on human health generated from primary pollutants ($SO_2$, CO, Particulates, VOCs) at distances up to 50 km from the emission source. Atmospheric modelling at local range was conducted by Roadpol, a Gaussian plume model. For greater distances more complex models are needed, especially for reactive pollutants, as atmospheric chemistry is important. Regional level damage assessment includes impacts on human health, materials and crops from primary and secondary (nitrates and sulphates aerosols, ozone) pollutants for the whole Europe, and was conducted by the Ecosense model. Global damage assessment includes impacts on human health and crops from ozone and impacts from global warming due to greenhouse gases ($CO_2$, $CH_4$, $N_2O$) emissions.

The external cost of transport for electric vehicles was estimated on the basis of the external cost of the electricity consumed by the vehicle, as electric vehicles have zero direct emissions but significant impacts are caused by electricity generation. Lignite power plants produce 76% of the electricity generated in the Greek mainland electricity system, oil power plants 14% and hydro power plants 10%. The damages per kWh for each power generation technology were estimated, using the damage estimates of the corresponding fuel cycles.

The damage estimates per building block for the "urban transport by car" case study are given in table 12.1. The most detrimental pollutant is $CO_2$ as it causes 44% of the total damage. Nitrate aerosols, VOCs, $SO_2$ and ozone share the rest of the damages.

**Table 12.1:** Damage estimates per building block for the "urban transport by car" case study, in mECU/vkm

| Location | Technology | Damages in mECU/vkm |
|---|---|---|
| Residential street | Passenger car* (cold engine) | 16.17 |
| Urban/residential street | Passenger car (warm engine) | 9.04 |
| Urban core principal street | Passenger car (warm engine) | 9.67 |
| Task average | | 10.93 |

* Unleaded petrol, capacity 1.4 l, fitted with closed loop catalytic converter

The damage estimates per building block for the „urban transport by car and electric bus" case study are given in table 12.2. The decrease of the task average damages, compared to those of table 12.1 are owed to the use of the trolley bus which has significantly lower damages from the passenger car.

**Table 12.2:** Damage estimates per building block for the "urban transport by car and electric bus" case study, in mECU/pkm

| Location | Technology | Damages in mECU/pkm |
|---|---|---|
| Residential street | Passenger car* (cold engine) | 16.17 |
| Urban/residential street | Passenger car (warm engine) | 9.04 |
| Urban core principal street | Trolley bus | 3.82 |
| Task average | | 8.28 |

* Unleaded petrol, capacity 1.4 l, fitted with closed loop catalytic converter

The damage estimates for the „regional transport by car" case study are given in table 12.3. The total damages have been reduced, compared to the damages of the urban transport task, and $CO_2$ damages represent 55% of the total damages as the local damages have been significantly reduced due to the lower population density of the local reference area.

**Table 12.3:** Damage estimates per building block for the "regional transport by car" case study, in mECU/vkm

| Location | Technology | Damages in mECU/vkm |
|---|---|---|
| Urban principal street | Passenger car* | 14.97 |
| Motorway 80-100 km/h | Passenger car | 8.43 |
| Task average | | 8.93 |

* Unleaded petrol, capacity 1.4 l, fitted with closed loop catalytic converter

The damage estimates for the „regional transport by truck" case study are given in table 12.4. Particulates are now the most detrimental pollutant as they represent 85% of the total damage. This results in high local range damage estimates, especially in the densely populated urban areas.

**Table 12.4:** Damage estimates per building block for the „regional transport by truck" case study, in mECU/vkm and mECU/tkm

| Location | Technology | Damages in mECU/vkm | Damages in mECU/tkm* |
|---|---|---|---|
| Urban principal street | Truck >16 t | 3515.1 | 305.7 |
| Motorway 80-100 km/h | Truck >16 t | 430.7 | 37.4 |
| Task average | | 662.3 | 57.6 |

* Average load of the truck: 11.5 tonnes

The damage estimates for the „regional transport by train" case study are given in table 12.5. Again particulates are the most detrimental pollutant, causing high impacts especially at urban densely populated areas. Despite the particulates impacts, the task average damages per pkm are lower from those of the passenger car (see table 12.5).

**Table 12.5:** Damage estimates per building block for the "regional transport by train" case study, in mECU/vkm and mECU/pkm

| Location | Technology | Damages in mECU/vkm | Damages in mECU/pkm* |
|---|---|---|---|
| Urban | Railcar | 18693.9 | 63.16 |
| Rural | Railcar | 1448.6 | 4.89 |
| Task average | | 1961.0 | 6.63 |

* Passenger capacity: 296 passengers

Except for the impacts caused by the atmospheric pollutants emitted during the transportation stage, impacts are also generated during other stages of the life cycle of the transport means such as production, maintenance and disposal of vehicles, fuel production and construction of the infrastructure (roads, railroads etc.). An attempt to quantify the emissions of those stages with an emphasis on fuel production, which is considered to be the one of the most significant contributors to the emissions caused by the above mentioned life cycle stages, was made.

The results show that there is a strong dependency of the damage estimates to the location of the transport tasks. More specifically local impacts, which represent a significant part of the total damages, appear to be very high in densely populated urban areas, while the regional impacts remain almost constant. Concerning differences between technologies, diesel vehicles appear to have much higher impacts than the petrol powered vehicles mainly due to their particulate emissions. The use of electric bus in urban areas proved to be very effective, concerning the reduction of damages, as they have zero direct emissions and do not affect their local environment. The cold start effect increases significantly the damage estimates, especially in urban areas where the trip lengths are short and a big proportion of the trip is driven with cold engine.

In general, for urban areas the trolley bus was found to have lower impacts per pkm compared to catalytic cars, despite the fact that electricity in Greece is produced by lignite. However the development of technologies that will reduce emissions at cold engine conditions, would cause a significant reduction to the impacts of passenger cars in urban areas. For rural areas, the diesel train was found to have lower impacts per pkm than the petrol car, despite the lack of an advanced emission control system.

# 13. CASE STUDY ITALY

## 13.1 Introduction

The negative effects of pollution and congestion given by urban traffic have been of serious and increasing concern in the last two decades. A case-study focused on a transport task within Milan appears to be particularly interesting because of the particular characteristics of the second largest town (in terms of population) in Italy. In fact, this is considered as one of the most polluted of the whole country. The location of the town - in the middle of Po valley - has not particularly favourable climatic conditions from this point of view. As a matter of fact, the relatively high temperature (ranging from an average of 4°C on January to about 25°C on July), the low mixing height and finally the absence of frequent and strong winds favour the persistence of primary pollutants in the atmosphere and the creation of secondary (photochemical) pollutants in the area considered. Moreover, the high population density and the high number of cars per inhabitant render the problems of traffic congestion and of people's exposure to respiratory illness particularly serious.

## 13.2 Transport task, localisation and technology

The case study considers private passenger transport between the town centre and the outskirts, as this transport task and localisation turn out be the most frequent in the metropolitan area of Milan. In particular, the analysis focuses on two main streets - viale Monza and via Ripamonti - that with their 4.2 and 5.2 km length link correspondingly the Northern and Southern part of the outskirts' ring with the historical centre of Milan. In both cases, the population density surrounding the street examined is considerable, with an average of about 8830 inhabitants per squared km.

The study analyses two different technologies well representative of the Italian reality: a petrol-fuelled, low displacement petrol car fitted with Three Way Catalyst (TWC), and a diesel car with a relatively low (less than 2,0 l.) engine displacement. Emission factors were estimated according to the /CORINAIR 1991/ report, and calibrated on traffic data (provided by the local Public Transport Utility, ATM) representative of the driving cycle of the street considered. The traffic data were taken on each street for both directions (IN = towards the town centre; OUT = towards the outskirts) and for different times of the day ("peak hours", from 7.30 to 8.30 am, and "off-peak hours", from 10 to 11 am).

**Table 13.1:** Traffic conditions on the streets analysed

| Street and time considered | Average Speed (km/h) | |
|---|---|---|
| | Way OUT | Way IN |
| Monza (peak h.) | 30,1 | 17,8 |
| Monza (off-peak h.) | 28,5 | 27,5 |
| Ripamonti (peak h.) | 33,0 | 30,9 |
| Ripamonti (off-peak h.) | 40,0 | 36,1 |

Both streets considered are particularly congested, with very low traffic average speed. Therefore, the cases chosen should rather be interpreted as "worst cases" for the environmental externalities of urban private transport.

## 13.3 Assessment of the environmental externalities

The externalities were evaluated according to the ExternE methodology, using a set of Exposure-Response Functions and Monetary Unit Values commonly defined by all the research teams involved in the program. The dispersion models used for atmospheric pollutants were ROADPOL, for the local scale (a 40x40 km square around the street, with 2 km step defined for the receptor density), and ECOSENSE for the regional one.

The very bulk of the external damage is due to atmospheric pollutants emitted by the vehicles, and in particular to the finer particulates ($PM_{2.5}$). This is the reason why the damage caused by the diesel vehicle appears to be much higher than that originated by the petrol car. The large difference found between the damages caused by diesel and petrol vehicle, however, should be read by keeping in mind to limits of the analysis. First, the evaluation does not take into account secondary pollutants (sulphates, nitrates and Ozone) originated by SOx, NOx and VOC at the local scale. Their quantification, if possible, would increase the damage estimate for petrol car relative to the diesel car, since the first vehicle has higher emission factors for the primary pollutants involved. Moreover, it is very likely that petrol cars have some particulate emissions as well, even if lower than the diesel-fuelled vehicles. This report does not consider any PM emission from petrol cars, and the final result for this technology is therefore correspondingly biased against diesel vehicles.

The damage due to all carcinogenic pollutants appears to be much lower - although not negligible -compared to that caused by particulates. It has to be emphasised, however, that the estimates regarding carcinogenic pollutants are affected by a much higher uncertainty, as there are difficulties in finding reliable ERFs and monetary unit values for these effects.

**Table 13.2:** Total damage of diesel and petrol vehicles(v.le Monza, Direction IN, peak hours)

| | **Diesel** (mECU/vkm) | **Petrol** (mECU/vkm) |
|---|---|---|
| Health (local) | 1600.7 | 3 |
| soiling (local) | 9.9 | 0 |
| agriculture (local) | 0.1 | 0.1 |
| Health (regional) | 13.50 | 3.26 |
| Others (regional) | 0.02 | 0.09 |
| *Total* | *1623.5* | *6.5* |

As shown in table 13.2, the monetary damage turns out to be considerably high. Although the estimates have to be taken with caution, they suggests that transport externalities are probably far from being fully internalised by any tax posed on them in Europe. It is interesting to notice that externalities due to the human health impact are much higher than those due to "global" problems, such as greenhouse and ozone effects.

**Table 13.3:** Global warming and ozone damage (v.le Monza, Direction IN, peak hours)

| | Diesel (mECU/km) | Petrol (mECU/km) |
|---|---|---|
| Global warming | 4.1 | 3.1 |
| Ozone | 0.02 | 0.03 |

The damage appears to be of a very local nature, i.e. about 80% of the total damage occurs within a 18x18 km square around the street (table 14.4). Even by extending the analysis on a 100x100 km square, the results do not show significant changes from those obtained on the 40x40 km area. This local nature renders the evaluation extremely site-specific, therefore particularly difficult to be extended to other cases.

**Table 13.4:** Variability of damage with the extension of the area considered (Case: Diesel vehicle, Monza-IN, peak hours)

| Area | Health damage (ECU/vkm) | % of total |
|---|---|---|
| 6x6 km | 0.803 | 50.1% |
| 18x18 km | 1.295 | 80.7% |
| 40x40 km | 1.601 | 99.8% |
| 100x100 km | 1.604 | 100.0% |

## 13.4 Damage due to car accidents

The Italian case-study also evaluated the damage due to car accidents, although this is of a different nature from the environmental externalities. A mortality index for car accidents in the urban area of Milan was estimated in order to achieve an evaluation of this effect. The index obtained was of 6.61 deaths per 100 millions vkm. This result is lower than the average value that the same index takes for different EU countries, as shown in table 13.5. This can be explained by the fact that our index only concerns urban traffic, where the average speed (and therefore the probability of having casualties from car accidents) is much lower than the average calculated for all kinds of trips.

The monetary unit value chosen to estimate this impact has been the VSL (Value of Statistical Life) of 3,1 millions ECU. This value has been preferred to the YOLL (Year Of Life Lost) since death in car accidents are likely to concern people with a relatively lower age then those dying by illness due to atmospheric pollution.

The final result obtained for this effect is of 205 mECU/km. This estimate can be considered of the same order of reliability of those presented for local mortality and morbidity. Its uncertainty depends mostly on the VSL estimate. This result also offers an useful term of comparison to judge the results found for environmental externalities.

**Table 13.5:** Mortality indices for road traffic accidents in EU countries, 1994

| Country | Mortality Index |
|---|---|
| Italy | 17.5 |
| Germany | 16.8 |
| United Kingdom | 8.8 |
| France | 18.5 |
| Netherlands | 12.0 |
| Austria | 21.0 |
| Switzerland | 11.3 |
| Denmark | 13.6 |
| Finland | 11.5 |
| EIRE | 15.8 |
| Iceland | 9.1 |
| USA | 10.7 |

Source: /ISTAT 1995/

# 14. CASE STUDIES NETHERLANDS

This study shows that externalities from transport related air emissions are much higher than estimates made in the past indicated. In this study the externalities of different passenger and goods transport tasks including different transport modes (road, rail and water transport) and different transport technologies (present and future) were different areas in the Netherlands. The transport tasks considered for both passenger and goods transport in these Dutch case studies are:

*Amsterdam to* Schiphol *airport (as)*
This transport task is 16.5 km for both road and rail transport and mainly crosses a densely populated area (6000-7000 people per square kilometre).

*Amsterdam Central Station to the* 'Vrije *Universiteit' (au)*
This transport task is 7.0 km for cars, 6.9 km for busses, 13.4 km for trains, 6.4 km for trams (passengers only) and 9.2 km for the metro (passengers only). It goes through a densely populated area (6000-7000 people per square kilometre).

*Rotterdam docks to Nijmegen docks (rn)*
This transport task is 145.3 km for road transport, 141.3 km for rail transport and 148.1 km for ship transport (goods only). It mainly goes through moderately to sparsely populated areas (200-3000 people per square km).

*Schiphol airport to Rotterdam Central Station (sr)*
This transport task is 60.1 km for road transport and 66.5 km for rail transport. It mainly crosses a moderately populated area (500-3000 people per square km).

In order to compare the externalities of different technologies and transport tasks it is useful to look at the relative contribution of pollutants to the total damage estimates. Results showed that most damages are due to emissions of NOx (40-80%), primary particles (45-15%), and $CO_2$ (10-15%) for petrol car. For a diesel car 80 to 90% of the damages are due to primary particulate emissions. NOx, $SO_2$ and $CO_2$ emission impacts all account for an impact of one or more mECU/vkm for both car types while primary particles emissions from diesel cars account for about ten times higher damages. NOx emissions mainly lead to regional range health impacts due to nitrate aerosols and ozone formed in the atmosphere. $SO_2$ emissions mainly lead to regional range health impacts due to sulphate aerosols formed in the atmosphere. Primary particle emissions mainly lead to direct local range health impacts.

**Table 14.1:** Passenger transport mid damage estimates in mECU/vkm, mECU/pkm and mECU/ptt. The uncertainty is of class B (a factor 4-6).

| Vehicle type | Fuel type | Technology | Cyl. Capacity | Year built | Damage in mECU/vkm | | | | Damage in mECU/pkm | | | | Damage in mECU/ptt | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | *as* | *au1* | *rn* | *sr* | *as* | *au1* | *rn* | *sr* | *as* | *au1* | *rn* | *sr* |
| Car | Petrol | No cat | 1.4-2.0 | 1990 built | 28.9 | 21.9 | 31.6 | 33.0 | 20.2 | 15.3 | 22.1 | 23.1 | 335 | 108 | 3217 | 1389 |
| | | TWC reg. | 1.4-2.0 | 1990 built | 7.7 | 7.6 | 7.3 | 7.7 | 5.4 | 5.3 | 5.1 | 5.4 | 90 | 38 | 738 | 324 |
| | | Average | Average | 1990 built | 11.3 | 9.4 | 11.8 | 12.4 | 7.9 | 6.6 | 8.3 | 8.7 | 131 | 47 | 1203 | 521 |
| | | TWC reg. | 1.4-2.0 | 2000 built | 4.8 | 5.2 | 4.2 | 4.5 | 3.4 | 3.6 | 2.9 | 3.1 | 56 | 26 | 426 | 188 |
| | Diesel | IDI | 1.4-2.0 | 1990 built | 77.4 | 83.8 | 54.5 | 68.1 | 54.2 | 58.6 | 38.2 | 47.7 | 896 | 413 | 5546 | 2866 |
| | | DI | 1.4-2.0 | 1990 built | 168.8 | 180.4 | 118.1 | 149.5 | 118.2 | 126.3 | 82.7 | 104.6 | 1954 | 889 | 12012 | 6291 |
| | | Average | Average | 1990 built | 79.5 | 86.0 | 56.0 | 69.9 | 55.6 | 60.2 | 39.2 | 48.9 | 920 | 424 | 5694 | 2941 |
| | | IDI | 1.4-2.0 | 2000 built | 39.2 | 41.7 | 28.4 | 35.1 | 27.4 | 29.2 | 19.9 | 24.5 | 454 | 205 | 2885 | 1476 |
| | LPG | No cat | 1.4-2.0 | 1990 built | 19.7 | 17.6 | 19.6 | 20.7 | 13.8 | 12.3 | 13.7 | 14.5 | 228 | 87 | 1992 | 870 |
| | | TWC reg. | 1.4-2.0 | 1990 built | 8.7 | 8.7 | 8.4 | 8.7 | 6.1 | 6.1 | 5.9 | 6.1 | 101 | 43 | 856 | 367 |
| | | Average | Average | 1990 built | 9.2 | 8.7 | 9.1 | 9.4 | 6.4 | 6.1 | 6.4 | 6.6 | 107 | 43 | 927 | 397 |
| | | TWC | 1.4-2.0 | 2000 built | 4.8 | 4.9 | 4.5 | 4.6 | 3.3 | 3.5 | 3.1 | 3.2 | 55 | 24 | 454 | 195 |
| | Average | Average | Average | 1990 built | 23.8 | 23.8 | 19.7 | 22.7 | 16.7 | 16.7 | 13.8 | 15.9 | 276 | 117 | 2002 | 956 |
| | Average | Average | Average | 1990 park | 44.9 | 49.4 | 35.5 | 40.1 | 31.4 | 34.6 | 24.8 | 28.0 | 520 | 245 | 3608 | 1686 |
| Bus | Diesel | Average | Average | 1990 built | 572 | 780 | 322 | 401 | 47.6 | 65.0 | 26.8 | 33.4 | 788 | 446 | 3898 | 2010 |
| | | Average | Average | 1990 park | 628 | 841 | 370 | 451 | 52.4 | 70.1 | 30.8 | 37.6 | 866 | 481 | 4477 | 2259 |
| | | Average | Average | 2000 built | 77 | 102 | 50 | 57 | 6.4 | 8.5 | 4.2 | 4.8 | 106 | 58 | 610 | 287 |
| Train | Electric | Stopping train | | | 152 | 152 | 152 | 152 | 2.4 | 2.4 | 2.4 | 2.4 | 40 | 33 | 344 | 162 |
| | | Inter-city train | | | 125 | 125 | 125 | 125 | 1.5 | 1.5 | 1.5 | 1.5 | 24 | 20 | 209 | 98 |
| | | High speed train | | | 299 | 299 | 299 | 299 | 1.1 | 1.1 | 1.1 | 1.1 | 18 | 15 | 156 | 74 |
| | | Average train | | | n.d. | n.d. | n.d. | n.d. | 1.7 | 1.7 | 1.7 | 1.7 | 28 | 23 | 241 | 114 |
| Tram | | Average | | 1995 | n.a. | 76 | n.a. | n.a. | n.a. | 3.9 | n.a. | n.a. | n.a. | 25 | n.a. | n.a. |
| Metro | | Average | | 1995 | n.a. | 114 | n.a. | n.a. | n.a. | 4.0 | n.a. | n.a. | n.a. | 37 | n.a. | n.a. |

* TWC reg. = regulated Three Way Catalyst, No cat = No Catalyst (Super petrol fuelled), IDI = Indirect injection, DI = direct injection, n.d. = no data available, n.a. = not analysed as transport mode is not available.

**Table 14.2:** Goods transport total mid damage estimates in mECU/vkm, mECU/tkm and mECU/ttt. The uncertainty is of class B (a factor 4-6).

| | | | | | Damage in mECU/vkm | | | | Damage in mECU/tk | | | | Damage in mECU/ttt | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Vehicle type | Fuel type | Technology | Cyl. Capacity | Year built | *as* | *au1* | *rn* | *sr* | *as* | *au1* | *rn* | *sr* | *as* | *au1* | *rn* | *sr* |
| Van | Petrol | No cat | 1.4-2.0 | 1990 built | *47.9* | *39.2* | *49.5* | *52.5* | *174* | *143* | *180* | *191* | *2882* | *1004* | *26138* | *11479* |
| | | TWC reg. | 1.4-2.0 | 2002 built | *9.2* | *9.2* | *8.2* | *9.0* | *34* | *33* | *30* | *33* | *554* | *235* | *4322* | *1962* |
| | Diesel | IDI | 1.4-2.0 | 1990 built | *119.2* | *126.1* | *86.8* | *107.1* | *434* | *459* | *316* | *389* | *7170* | *3225* | *45860* | *23407* |
| | | DI | 1.4-2.0 | 1990 built | *151.3* | *186.9* | *98.7* | *118.7* | *550* | *679* | *359* | *432* | *9097* | *4780* | *52130* | *25954* |
| | | IDI | 1.4-2.0 | 2002 built | *28.6* | *29.3* | *22.9* | *26.7* | *104* | *107* | *83* | *97* | *1717* | *749* | *12087* | *5834* |
| | LPG | No cat | 1.4-2.0 | 1990 built | *34.1* | *22.6* | *41.0* | *41.5* | *124* | *82* | *149* | *151* | *2050* | *579* | *21639* | *9068* |
| | | TWC reg. | 1.4-2.0 | 2002 built | *13.2* | *12.2* | *12.8* | *13.6* | *48* | *44* | *47* | *50* | *796* | *313* | *6757* | *2981* |
| | Average | Average | Average | 1990 built | *96.4* | *98.4* | *74.8* | *89.6* | *351* | *358* | *272* | *326* | *5800* | *2516* | *39494* | *19588* |
| Truck | Diesel | Average | 3.5-16 t | 1990 built | *330* | *361* | *241* | *290* | *117* | *128* | *85* | *103* | *1929* | *898* | *12364* | *6166* |
| | | Average | >16 t | 1990 built | *534* | *660* | *345* | *420* | *80* | *99* | *52* | *63* | *1320* | *693* | *7483* | *3777* |
| | | Average | Average | 1990 built | *483* | *585* | *319* | *388* | *84* | *102* | *56* | *68* | *1396* | *719* | *8086* | *4072* |
| Puller | Diesel | Average | > 16 t | 1990 built | *761* | *989* | *462* | *566* | *76* | *98* | *46* | *56* | *1253* | *693* | *6681* | *3390* |
| | | Average | Average | 1990 built | *576* | *721* | *367* | *448* | *80* | *100* | *51* | *62* | *1328* | *706* | *7425* | *3751* |
| | | Average | Average | 1998 built | *72* | *86* | *54* | *60* | *10.1* | *12.0* | *7.5* | *8.4* | *166* | *85* | *1094* | *504* |
| Train | Diesel | DE64000 | | 1996 built | *n.d.* | *n.a.* | *n.d.* | *n.d.* | *9.1* | *n.a.* | *8.0* | *8.8* | *150* | *n.a.* | *1127* | *585* |
| | | DE1024 | | 2010 built | *n.d.* | *n.a.* | *n.d.* | *n.d.* | *7.6* | *n.a.* | *6.6* | *7.3* | *126* | *n.a.* | *927* | *488* |
| Train | Electric | 1.5 kV | | 1990 built | *n.d.* | *n.a.* | *n.d.* | *n.d.* | *0.8* | *n.a.* | *0.8* | *0.8* | *12* | *n.a.* | *106* | *50* |
| | | 25 kV | | 2000 built | *n.d.* | *n.a.* | *n.d.* | *n.d.* | *0.8* | *n.a.* | *0.8* | *0.8* | *13* | *n.a.* | *109* | *51* |
| Ship | Diesel | Small | | 1990 fleet | *n.a.* | *n.a.* | *n.d.* | *n.a.* | *n.a.* | *n.a.* | *2.8* | *n.a.* | *n.a.* | *n.a.* | *419* | *n.a.* |
| | | Medium | | 1990 fleet | *n.a.* | *n.a.* | *n.d.* | *n.a.* | *n.a.* | *n.a.* | *3.2* | *n.a.* | *n.a.* | *n.a.* | *475* | *n.a.* |
| | | Push vessel | | 1990 fleet | *n.a.* | *n.a.* | *n.d.* | *n.a.* | *n.a.* | *n.a.* | *4.0* | *n.a.* | *n.a.* | *n.a.* | *594* | *n.a.* |

* TWC reg. = regulated Three Way Catalyst, No cat = No Catalyst (Super petrol fuelled), IDI = Indirect injection, DI = direct injection, n.d. = no data available, n.a. = not analysed as transport mode is not available.

Table 14.1 and table 14.2 show, for passenger and goods transport respectively, the total core externality estimates. The externalities are reported as damage costs in mECU per vehicle kilometre (vkm), passenger or tonne kilometre (pkm or tkm) and passenger transport task (ptt) or tonne transport task (ttt).

## 14.1 Present technologies

It can be concluded that for 1990 transport technologies, the electric rail transport modes (i.e. trains and for passenger transport also tram and metro) are preferred over all fossil fuelled transport modes on a passenger or tonne kilometre and passenger or tonne transport task basis. This is because emissions from fossil fuelled road transport technologies occur at the road side while emissions from electricity production takes place from high stacks. Electricity production leads to relatively low local concentrations and therefore low local impacts. Emission at the road side lead to relatively high concentrations near roads, resulting in high local impacts. These local impacts mainly determine the total damages for fossil fuelled road traffic. As regional impacts only increase moderately with stack height, the balance for the total externalities favours electric vehicles over fossil fuelled vehicles. The transport emissions from fossil fuelled vehicles are also much higher on a passenger or tonne kilometre basis.

Furthermore, for passenger road transport a petrol or LPG fuelled car is preferred over a diesel bus, and the diesel bus is in its turn preferred over a diesel car on a passenger kilometre and transport task basis. The reason for diesel busses having such high externalities is due to very high particulate matter emissions. So, even though the average capacity use of busses is much higher than of cars, 12 p/vkm and 1.43 p/vkm respectively, busses score very poorly.

For goods road transport puller and large truck transport are preferred over small truck and diesel van transport on tonne kilometre and transport task basis. Especially diesel vans have very high externalities. The difference is due to the fact that the average trip load of vans (0.275 t/vkm) and small trucks (5.8 t/vkm) are small compared to the large trucks (13.7 t/vkm) and pullers (22.3 t/vkm). The diesel trains, but also the electric trains, have relatively low externalities per tkm due to the high average load (522 t) per vkm.

The preferences given above are not valid if the comparisons are based on externalities per vehicle kilometre (vkm). However, this is, from a transport task performance perspective, not the right comparison because also the distance travelled with each transport mode has to be taken into account.

## 14.2 Future technologies

Due to restrictions on emissions from fossil fuelled vehicles and corresponding technology changes, impacts for some future transport technologies will be reduced considerably. High future reductions in externalities are especially observed with diesel bus technologies. This observation is due to the fact that diesel busses in 1990 were mainly based on old technology with high emissions. This development will bring externalities from a passenger transport task with diesel busses closer to those with clean petrol and LPG cars. Using natural gas fuelled busses (which have much lower emissions) and electric busses would probably lower the externalities even more. High reductions in externalities are especially attainable with future diesel truck and puller technologies. This is due to the fact that in 1990 they were mainly based on old technology with high emissions. This development will make externalities from goods transport task with these transport modes comparable to those with diesel trains. Electric train transport (including diesel trains) will, however, also in the future, give lowest externalities. This is especially the case when the efficiency of train transport will be improved and electricity use reduced. However, with the foreseen technological developments ship transport externalities will probably be reduced significantly. It is possible that ship externalities could become as low as electric train externalities.

## 14.3 Drive type influence on the externalities

For a simple aggregation, from individual car externalities to a whole traffic scheme or even a whole transport fleet, average externality estimates have to be obtained. For electric vehicles this is relatively simple as the impacts from electricity production are only due to emissions from a limited number of electricity plants. The results from this study show that externalities from fossil fuelled road transport are highly depending on the population density and the transport technology. The type of drive (urban, other road or highway) and the location of the drive are of less influence on the damage. It seems that in sparsely populated areas the results for the different drive types converge, which is not surprising. For the same reasons these trends are also observed for goods transport. From the results no statistically justifiable average externalities for technologies in lowly, moderately and highly populated areas could be derived. More transport tasks in more locations would have to be studied.

## 14.4 Sensitivity analyses

The influence of several parameters, such as the valuation methods, treating particle emissions from transport as PM10 instead of as $PM_{2.5}$, additional impact functions, and the inclusion of all life cycle stages, on the externalities were considered. The results indicate that:

- Changing the valuation method from a 'years of life lost' approach to a 'value of statistical life' approach resulted in increases in the estimates of roughly 110-270% depending on the

transport technology. The chronic health impacts alone are increased by roughly a factor 4 by this change in valuation.

- Treating particle emissions from transport as PM10 instead of as $PM_{2.5}$ results in a decrease of the damage estimates of 4 to 36%. The decrease is smallest for petrol and LPG cars as they have relatively low particle emissions.
- Including additional impacts exposure-response functions, on which health experts are still in discussion concerning the causality of the impact to the pollutants, did only increase the impacts by 1-15%.

Because the externalities of the other life cycle stages (i.e. vehicle production, maintenance and disposal, infrastructure construction and maintenance and fuel production) are hard to quantify, they were not included in the 'core' estimates. The provisional externality estimates for these stages for passenger transport would be 8.1, 2.5, 2.0, 5.1, 4.2, 4.2 and 3.0 mECU/pkm for cars, stopping trains, inter-city trains, high speed trains, trams, metros and busses respectively. For goods transport this would be roughly, 49.6, 10.0, 6.3, 6.2, 2.7 and 4.9 mECU/tkm for vans, small trucks, large trucks, pullers, electric trains and diesel trains respectively. For ships no data were available. The life cycle stage externalities lead to a significant increase in the externalities for train transport (an increase of a factor 1 to 5) and for petrol and LPG cars (roughly a doubling). Also for future technology diesel busses the externalities would increase by almost 50%. However, including these externalities for the present technologies would not change the preferences for the transport modes given before. For future technologies the life cycle stage impacts would give a diesel bus a slight preference over a petrol or LPG car. High speed trains would no longer be preferred over stopping trains and inter-city trains due to high externalities associated with the construction of high speed lines and the relatively low occupancy rate for the new infrastructure.

## 14.5 Comparison with results from other studies

How do the air pollution externality estimates of this study compare to the externality estimates obtained from studies looking at marginal costs of emission reduction - MCER - and studies using a top-down approach - TD ? The externality estimates from these studies are shown in table 14.3.

Though a straightforward comparison of the results is not possible some conclusions can be drawn. First of all it is clear that the summed estimates for road and water transport from this study are at least one order of magnitude higher than for both other studies. For rail transport the estimates from this study are lower than for the MCER study but higher than for the TD study. When considering that the global warming damage estimates in the this study and the MCER study are of the same order of magnitude, the large differences have to be found in the other air pollution impacts. In the MCER and TD studies only the pollutants NOx and VOC were considered and the damages due to these pollutants were estimated by studying direct impacts only. In this study NOx emissions were shown rather to cause high indirect impacts due to ozone and aerosol formation. Other pollutants that caused high damages in this study were primary particles, sulphate aerosols from $SO_2$ and to a lesser extent $SO_2$ itself. The large

differences can probably also be explained by the completely different approaches to quantifying and valuing the impacts in the studies.

**Table 14.3:** Externality estimates from the MCER and the TD approaches in mECU/tkm or mECU/pkm.

| | | MCER study [a] | | TD study [b] | |
|---|---|---|---|---|---|
| | Fuel type | Air pollution impacts | Global warming impacts | Air Pollution impacts | Global warming impacts |
| Goods transport technologies | | | | | |
| Van | average | - | - | 1.4 | - |
| Truck - small | diesel | - | - | 1.1 | - |
| Truck - large | diesel | 5.6 | 2.5 | 0.7 | - |
| Train | electricity | 0.8 | 2.0 | 1.1 | - |
| Ship | average | 6.0 | 0.6 | 0.25 | - |
| Passenger transport technologies | | | | | |
| Car | petrol | | | 1.0 [c] | - |
| | diesel | | | 0.6 [c] | - |
| | Average | 14.6 | 4.5 | | - |
| Train | electricity | 0.8 | 2.2 | 1.2 [c] | - |

[a] Source : Kaegeson, Getting the prices right, T&E, Stockholm, 1993

[b] Source: Boneschanker and t'Hoen, 'Externalities of goods transport, IOO, Den Haag , 1993

[c] These estimated are obtained by extrapolation on the basis of emission factors compared to goods transport modes.

# 15. CASE STUDIES UNITED KINGDOM

The UK case studies were split into two main assessments. The first was a comparison of the external costs of private and public transport in London and the second an evaluation of the external costs of goods transport and alternative fuel buses in Barnsley. A summary of these case studies is presented below.

## 15.1 Case Study I (London): An Analysis of Alternative Public and Private Transport Modes in Urban Areas

### 15.1.1 Reference Technologies and Location

The reference location was an urban transport corridor along the Edgware Road in London. The stretch of road considered was 11 km long, connecting central London to the north of the city. The study considered the following transport technologies:

Private car (average private car, based on the emission factors and the numbers in the vehicle fleet of small, medium and large cars, the proportion of pre-EURO, EURO I and EURO II vehicles, and the numbers of petrol and diesel cars);
Diesel bus with a EURO II engine and oxidation catalyst running on city-diesel fuel;
Electric bus;
Light-rail system.

### 15.1.2 Emission Factors

The data for the average private car was derived from aggregating the emission factors for each component of the vehicle stock. The car emission calculations were estimated using an average speed model, based largely on the CORINAIR methodology. Average speed emission functions were estimated for each vehicle category for fuel consumption and the regulated pollutants (CO, $NO_x$, VOCs, and PM). $CO_2$ and $SO_2$ emissions were calculated from a mass balance and from the mass of sulphur, together with fuel consumption. The average speed for the study was assumed to be 28 km/h. The emissions were adjusted to account for cold start effects, based on the CORINAIR methodology. The resulting emission factors are shown in table 15.1.

**Table 15.1:** Emission Factor for an Average Private Car in London

| Emission (g/km) | | | | | |
|---|---|---|---|---|---|
| $CO_2$ | CO | VOC | $NO_x$ | $SO_X$[1] | PM |
| 200 | 5.83 | 0.73 | 0.56 | 0.03 | 0.032 |

[1] Assumes fuel sulphur content of 0.02% for petrol and 0.05 for diesel.

The power station emissions associated with the electric vehicles were calculated assuming consumption rates of 1.4 kWh/km for the electric bus and 5.5 kWh/km for the light-rail system. The study assumed the electricity was supplied by the UK National Grid (year 2000). An additional sensitivity analysis was undertaken assuming the electricity was supplied by a marginal CCGT plant or modern-build coal plant (with FGD). The emission factors are shown in table 15.2.

**Table 15.2:** Emission Factors for the Electric Vehicles

| Intermediate | Generating | Emissions g/km | | | | | |
|---|---|---|---|---|---|---|---|
| Mode | Option | $CO_2$ | CO | VOC | $NO_x$ | $SO_x$ | PM |
| Light Rail | Average fuel mix | 2,351 | 0.35 | 0.22 | 5.45 | 9.46 | 0.55 |
| | Coal + FGD | 5,290 | 0.73 | 0.18 | 15.61 | 6.82 | 0.98 |
| | CCGT | 2,388 | 0.38 | 0.46 | 2.54 | 0.00 | 0.00 |
| Electric Bus | Average fuel mix | 598 | 0.09 | 0.06 | 1.39 | 2.41 | 0.14 |
| | Coal + FGD | 1,347 | 0.19 | 0.05 | 3.97 | 1.73 | 0.25 |
| | CCGT | 608 | 0.10 | 0.12 | 0.65 | 0.00 | 0.00 |

Source: /ETSU 1995a/.

### 15.1.3 Modelling Studies

The analysis of local air pollution dispersion was calculated for the car and bus emissions using the ROADPOL model (using historical 10 year average meteorological data). The ground level concentrations were predicted over a 50 $km^2$ area, using a 1 $km^2$ grid resolution; finer resolution was used close to the road. For regional pollution dispersion (both car and bus emissions and power station emissions) the EcoSense model was used.

### 15.1.4 Results

A summary of the results is presented in table 15.3.

**Table 15.3:** External Costs (mECU/km and /passenger km) from Transport Modes in London.

| Category | Local Impacts | Regional Impacts | Global Warming | Total (mECU/km) | Total (mECU/pkm) |
|---|---|---|---|---|---|
| Average car | 43.3 | 11.61 | 2.31 | 57.22 | 47.68 |
| Average bus | 553.2 | 127.4 | 14.54 | 695.14 | 57.93 |
| Modern bus | 208.6 | 155.2 | 21.50 | 285.30 | 32.11 |
| Electric bus | - | 41.6 | 8.35 | 49.95 | 4.16 |
| Light-rail | - | 164.3 | 33.10 | 197.40 | 4.94 |

The sensitivity analysis of electricity generation showed significantly lower damages per km when supplied by the CCGT plant. As an additional analysis, the net benefits from the introduction of the electric public transport systems was calculated. This involved estimating the level of modal shift from private cars to public transport along the transport corridor. The total net 'air pollution' benefits, from the reduction in local air pollution minus the impacts from electricity generation emissions, were calculated to be ECU 430,000 and ECU 270,000 per year for the light-rail and electric bus systems respectively. These results show the benefits of introducing such modes into urban areas.

## 15.2 UK Case study II (Barnsley): An Analysis of Urban Goods Vehicles and Alternative Fuel Buses

### 15.2.1 Reference Technologies and Location

The reference location was an urban transport corridor in Barnsley, a medium sized town in the north of England. The technologies considered included private cars, light and heavy goods vehicles and alternative fuel midi-buses (small buses). The technologies considered are shown in table 15.4.

### 15.2.2 Emission Factors

The emission calculations were estimated using the average speed approach. In order to estimate emission factors for the EURO II goods vehicles and the alternative fuel buses, scale factors were developed. The average speed for the case study was assumed to be 26 km/h. The values for the electric bus were estimated using consumption values of 1.4kWh/km and assuming power was supplied by marginal CCGT or Coal (FGD) plant. The emission factors are shown in table 15.4

**Table 15.4** Emission Factors for Reference Technologies

| | Emissions (g/km) | | | | | |
|---|---|---|---|---|---|---|
| VEHICLE TYPE | $CO_2$ | CO | VOC | $NO_x$ | $SO_x$ | PM |
| Car Petrol EURO I | 247.4 | 5.70 | 0.78 | 0.46 | 0.033 | 0.010 |
| Car Diesel EURO I | 245.2 | 1.08 | 0.24 | 0.47 | 0.079 | 0.229 |
| LGV Petrol EURO I | 372.3 | 7.5 | 0.43 | 0.42 | 0.05 | 0.02 |
| LGV Diesel EURO I | 312.4 | 2.0 | 0.20 | 0.78 | 0.1 | 0.28 |
| HGV Diesel <17T Pre EURO I | 616.9 | 3.67 | 2.49 | 5.77 | 0.20 | 1.53 |
| HGV Diesel <17T EURO I | 586.1 | 3.67 | 1.15 | 3.23 | 0.19 | 0.70 |
| HGV Diesel <17T EURO II | 556.8 | 3.26 | 1.15 | 2.84 | 0.18 | 0.18 |
| HGV Diesel >17T Pre EURO I | 1153.4 | 5.02 | 2.52 | 16.11 | 0.39 | 1.52 |
| HGV Diesel >17T EURO I | 1095.7 | 5.02 | 1.16 | 9.02 | 0.37 | 0.70 |
| HGV Diesel >17T EURO II | 1040.9 | 4.46 | 1.16 | 7.94 | 0.35 | 0.29 |
| Small Bus Diesel | 501.1 | 7.04 | 0.49 | 3.34 | 0.16 | 0.56 |
| Small Bus LPG | 476.0 | 3.31 | 0.40 | 0.60 | 0.00 | 0.28 |
| Small Bus CNG | 451.0 | 7.04 | 0.74 | 0.50 | 0.00 | 0.08 |
| Small Bus Methanol | (401.8) [1] | 4.74 | 0.07 | 1.65 | 0.00 | 0.03 |
| Small Bus Ethanol | (412.4) [1] | 6.26 | 0.17 | 3.45 | 0.00 | 0.03 |
| Small Bus Biodiesel | (511.1) [1] | 6.13 | 0.39 | 3.84 | 0.00 | 0.47 |
| Small Bus Electric | 0 | 0 | 0 | 0 | 0 | 0 |

[1] Biofuels can be assumed to be $CO_2$ neutral.

Source: /ETSU 1996/ Alternative Road Transport Fuels; /ETSU 1994/. UK Petrol and Diesel Study. LGV particulate figure taken from /ETSU 1994/ and assume pre-stage I vehicles. HGV particulate emissions based on /ETSU 1994/ urban factors and scale factors for stage I/II emissions. Bus particulate and $NO_x$ emissions based on /ETSU 1994/.

For the alternative fuel midi-buses, the emissions from other stages of the life cycle were calculated using the values derived within the LCA task of the project (/Collings 1996/; /ETSU 1995b/).

### 15.2.3 Modelling Studies

The analysis of local air pollution dispersion was calculated for the car and bus emissions using the ROADPOL model (using historical 10 year average meteorological data). The ground level concentrations were predicted over a 50 $km^2$ area, using a 1 $km^2$ grid resolution; finer resolution was used close to the road. For regional pollution dispersion (both car and bus emissions and power station emissions) the EcoSense model was used.

### 15.2.4 Results

A summary of the results are presented below in table 15.5, table 15.6 and table 15.7. Note for alternative fuel midi-buses, the results shown refer to the life cycle emissions.

**Table 15.5:** External Costs (mECU/km and /passenger km) from Private Cars (Barnsley).

| Category | Local Impacts | Regional Impacts[1] | Global Warming[1] | Total (mECU/km) | Total (mECU/pkm) |
|---|---|---|---|---|---|
| Petrol car | 4.03 | 7.69 | 2.90 | 14.62 | 12.18 |
| Diesel car | 93.3 | 9.46 | 2.87 | 105.63 | 88.03 |

[1] Impacts from end-use emissions only.

**Table 15.6:** External Costs (mECU/km and /tonne km) from Goods Vehicles (Barnsley).

| Category | Local Impacts | Regional Impacts[1] | Global Warming[1] | Total (mECU/km) | Total (mECU/tkm) |
|---|---|---|---|---|---|
| LGV petrol | 8.25 | 8.23 | 5.23 | 21.7 | 18.09 |
| LGV diesel | 114.2 | 13.55 | 4.39 | 132.1 | 110.12 |
| HGV<17 + pre EURO | 624.3 | 80.8 | 8.67 | 713.8 | 62.07 |
| HGV<17 + EURO I | 287.2 | 43.8 | 7.82 | 338.8 | 29.46 |
| HGV <17 + EURO II | 71.9 | 30.8 | 15.40 | 118.1 | 10.27 |
| HGV > 17 + pre EURO | 620.2 | 167.6 | 8.24 | 796.0 | 34.61 |
| HGV > 17 + EURO I | 285.4 | 93.4 | 16.21 | 395.0 | 17.17 |
| HGV > 17 + EURO II | 120.0 | 76.8 | 14.63 | 211.4 | 9.19 |

[1] Impacts from end-use emissions only.

**Table 15.7:** External Costs (mECU/km and /passenger km) from Alternative Buses (Barnsley)

| Midi-bus Category | Local Impacts | Regional Impacts[2] | Global Warming[2] | Total (mECU/km) | Total (mECU/pkm) |
|---|---|---|---|---|---|
| Diesel | 228.4 | 65.5 | 8.71 | 302.61 | 25.22 |
| LPG | 114.2 | 32.7 | 8.20 | 155.1 | 12.93 |
| CNG | 34.4 | 29.9 | 7.98 | 72.3 | 6.02 |
| Biomethanol | 13.3 | 40.0 | 2.71 | 56.01 | 4.67 |
| Bioethanol | 13.3 | 64.8 | 3.84 | 81.94 | 6.83 |
| Biodiesel | 191.9 | 75.6 | 3.15 | 270.65 | 22.55 |
| Electric (Coal+FGD) | - | 59.6 | 9.28 | 68.88 | 5.74 |
| Electric (CCGT) | - | 28.7 | 9.63 | 38.33 | 3.19 |

[2] Values include both end-use and other life-cycle stages.

## 15.3 Conclusions

The results from the two case studies provide an interesting comparison of the external costs of air pollution from passenger and goods transport in urban areas. The results of the assessment show the clear benefits of public transport modes, as the damages per passenger

km are much lower compared to private car trips. The benefits are greatest for electric systems, though modern diesel buses also show significant benefits.

The comparison of alternative urban goods in Barnsley shows the reduced external costs and therefore the benefits to air quality from introducing stricter EURO legislation. The results of the comparison of alternative fuels shows diesel and bio-diesel vehicles have the highest total impacts on air pollution, primarily because of the local impacts from particulate emissions, but also because of the regional impacts from high $NO_x$ emissions. These results show the potential benefits to air quality from introducing most alternative fuels in urban areas.

# 16. SUMMARY

From the case studies considered it can be concluded that health impacts dominate the damages quantified in this study; in particular, mortality due to primary ($PM_{2.5}$) and secondary particulates (nitrates, sulphates). Carcinogens, which were expected to play an important role due to their high specific toxicity, proved to be of much lower importance compared to the particles. The reason is, that carcinogens are only emitted in small quantities.

It was found, that the population density around a road is a key parameter for the magnitude of impacts, particularly for diesel vehicles. This effect is caused mainly by the importance of the primary particles for the total damage. Based on the case studies, three main categories of locations could be identified:
agglomerations (areas of highest population densities over a large area) e.g. Paris: ca. 10 million people on 50 x 50 $km^2$,
urban areas (high population density over a large area), e.g. Amsterdam, Barnsley, Stuttgart: ca. 2 million people on 50 x 50 $km^2$,
extra-urban areas (medium to low population densities).

Table 16.1 illustrates the damage due to a diesel car in selected locations. The clear distinction between the three location categories is obvious, having a look at the total results: 560 mECU/vehicle kilometre (vkm) for the agglomeration Paris, 65 to 107 mECU/vkm for the urban areas and 30 to 38 mECU/vkm for the extra-urban areas. The considerable differences, above all the high value for Paris, are mainly caused by the population densities around the road.

For comparison to the user costs: at a fuel consumption of 7 litres diesel per 100 km and a diesel price of ca. 600 mECU/litre /average price in Germany 1995 including fuel tax and VAT/, the fuel cost per vkm are 42 mECU/vkm. This means, that the quantified damages due to airborne pollutants and greenhouse gases reach at least the same order of magnitude as the fuel costs for a diesel vehicle.

At this stage of the project it is not possible to make a detailed comparison between countries. For instance, the range and level of detail of data available (e.g. emission factors) was not the same for the different case studies. The main source of differences between the single case study results is differing emission factors, illustrated in table 16.3. If the results are adjusted to the emissions of the German car (i.e. assuming that a German car runs in the different locations) the results for the different categories are: agglomeration 400 mECU/vkm, urban areas 50 to 65 mECU/vkm and extra-urban areas 20 to 30 mECU/vkm. This gives a clearer picture of the different location categories.

**Table 16.1:** Damage estimates (vehicle use only) for diesel passenger cars in different locations, given as 'best estimate' in mECU/vkm.

| Diesel car | agglomerations | urban areas | | | extra-urban areas (motorway drive) | | uncertainty label[1] |
|---|---|---|---|---|---|---|---|
| [mECU/vkm] | Paris | Stuttgart | Amsterdam | Barnsley | Stuttgart-Mannheim | Tiel drive | |
| | (FR) | (DE) | (NL) | (UK) | (DE) | (NL) | |
| Primary pollutants | | | | | | | |
| Particles ($PM_{2.5}$) | 534.09 | 50.43 | 78.60 | 97.40 | 18.77 | 29.50 | B |
| $SO_2$ | 0.93 | 1.12 | 0.71 | 0.80 | 0.60 | 0.32 | A/B |
| CO | 0.02 | 0.003 | 0.003 | 0.005 | 0.001 | 0.0004 | B |
| Cancers | 4.02 | 0.54 | 0.57 | 1.25 | 0.18 | 0.22 | B |
| Secondary pollutants | | | | | | | |
| Sulphates | 0.59 | 0.82 | 1.30 | 0.63 | 0.68 | 1.10 | B |
| Nitrates | 18.18 | 9.14 | 2.70 | 2.82 | 7.24 | 3.80 | B? |
| Ozone | 1.29 | 0.96 | 0.90 | 0.93 | 0.78 | 1.20 | B |
| Global warming | 2.97 | 2.28 | 2.70 | 3.45 | 1.99 | 2.30 | C |
| TOTAL | 562.1 | 65.3 | 87.5 | 107.3 | 30.2 | 38.4 | |

[1] A = high confidence (a factor 2.5 to 4); B = medium confidence (a factor 4 to 6); C = low confidence (a factor 6 to 12); „?" = evidence is weak

**Table 16.2:** Damage estimates (vehicle use only) for 'three-way-catalyst' passenger cars in different locations, given as 'best estimate' in mECU/vkm.

| TWC car | agglomerations | | | urban areas | | | extra-urban areas (motorway drive) | | uncertainty label[1] |
|---|---|---|---|---|---|---|---|---|---|
| [mECU/vkm] | Paris (FR) | Athens (GR) | Milan (IT) | Stuttgart (DE) | Amsterdam (NL) | Barnsley (UK) | Stuttgart-Mannheim (DE) | Tiel drive (NL) | |
| Primary pollutants | | | | | | | | | |
| Particles ($PM_{2.5}$) | 53.41 | n.q. | n.q. | 3.73 | 1.96 | 4.17 | 1.10 | 0.74 | B |
| $SO_2$ | 1.05 | 1.44 | 3.05 | 0.12 | 0.10 | 0.33 | 0.06 | 0.07 | A/B |
| CO | 0.06 | 0.04 | 0.13 | 0.02 | 0.02 | 0.025 | 0.004 | 0.001 | B |
| Cancers | 0.33 | 1.75 | 1.01 | 0.18 | 0.06 | 0.76 | 0.03 | 0.06 | B |
| Secondary pollutants | | | | | | | | | |
| Sulphates | 0.66 | 0.22 | 0.43 | 0.09 | 0.21 | 0.26 | 0.07 | 0.22 | B |
| Nitrates | 16.14 | 1.62 | 2.78 | 4.58 | 1.60 | 2.76 | 5.89 | 2.59 | B? |
| Ozone | 1.24 | 1.11 | 0.03 | 0.54 | 0.51 | 1.42 | 0.64 | 0.83 | B |
| Global warming | 3.58 | 4.73 | 5.40 | 2.98 | 3.20 | 3.48 | 2.38 | 2.49 | C |
| TOTAL | 76.5 | 10.9[2] | 12.8[2] | 12.2 | 7.7 | 13.2 | 10.2 | 7.0 | |

[1] A = high confidence (a factor 2.5 to 4); B = medium confidence (a factor 4 to 6); C = low confidence (a factor 6 to 12); „?" = evidence is weak .

[2] total includes no primary particles

technologies the life-cycle impacts are dominated by far by infrastructure production, maintenance and disposal, which cause at least 90% of the life-cycle damages.

**Table 16.4:** Damages due to up- and downstream processes of passenger transport technologies

| Technology | Damage in mECU/pkm | Country |
|---|---|---|
| Petrol car | 9.0 | DE |
| Diesel car | 7.1 | DE |
| Average car | 8.1 | NL |
| Diesel bus | 1.0 - 3.0 | NL - DE |
| Local train (electric) | 2.5 - 5.5 | NL - DE |
| Inter-city train (electric) | 2.0 - 2.4 | NL - DE |
| High speed train (electric) | 5.1 - 5.5 | NL - DE |

For goods transport, the advantage of trains as regards the damages due to operation is higher than for passenger transport. While goods trains cause quantifiable damages of 0.8 to 8.8 mECU/tonne kilometre (tkm) for electric and diesel traction, the damages for road transport ranges from 33 to more than 400 mECU/tkm. The low estimate refers to a petrol van complying with EURO2, the upper value to a diesel van - 1990ies technology. Heavy goods vehicles lie in-between. Even though the damages *per vehicle kilometre* rise with the size of the lorries, the damages *per tonne kilometre* decrease, as the load capacity (and average load) rises more than the damages.

The quantified damages are clearly dominated by mortality impacts, in particular by mortality due to particles. To illustrate the sensitivity to some key assumptions, table 16.5presents the variation of results for three example technologies: a diesel car, a petrol car fitted with three-way catalyst and an electric passenger train.

**Table 16.5:** Sensitivity of results to key assumptions (see text). Results are taken from German case study Stuttgart - Mannheim trip.

| mECU/pkm | standard ERF | | Sensitivity (0.5 * /Pope et al. 1995/) | |
|---|---|---|---|---|
| | VLYL | VSL | VLYL | VSL |
| Diesel car | 20.7 | 66.2 | 12.8 | 40.3 |
| Petrol car - TWC | 7.0 | 20.3 | 4.9 | 13.3 |
| Inter-city train | 2.2 | 4.2 | 2.0 | 3.5 |

ERF = exposure-response function; VLYL = value of a life year lost approach; VSL = value of a statistical life approach; TWC = three-way catalyst; pkm = passenger kilometre.

One key assumption is the use of the VLYL approach (value of a life year lost), based on the quantification of the number of years of life lost (YOLL) due to mortality. In the past, valuation in the ExternE Project was based on the VSL approach (value of a statistical life), not related to the years of life lost. The life time lost per premature death is estimated to 0.75

years for acute mortality and to 11 years for chronic mortality. Assuming a constant VLYL for every year (but including discounting of future life years), the both approaches lead to considerable differences.

The second sensitivity which is illustrated in table 16.5 appears in the context of transferring the results of a cohort study on mortality due to particles /Pope et al. 1995/ from the US to Europe. Studies on acute mortality in the US lead to results, which were a factor of two higher than comparable studies in Europe. The reason may be that the mass of particles, which was used as indicator, is not the appropriate indicator. The composition with respect to size and toxicity of particles is supposed to be different in the US and Europe. Table 16.5 shows the results for the sensitivity assumption that the slope of the corresponding exposure-response function is divided by two for a transfer to Europe. As the exposure-response function in question makes a considerable part of the total damages for road transport vehicles with internal combustion engine, this assumption has a considerable effect on the results. For the electric train the effect is much smaller, as global warming damages dominate the result.

Sensitivity analysis with other exposure-response functions suggested as "sensitivity" functions (see section on health effects) do not have a significant effect on the total damage /Rabl and Spadaro 1997/.

**Table 16.6:** Global warming damages due to $CO_2$ based on different damage factors.

| Vehicle[1] | IPCC range (5 - 125 $/t C) (1 - 30 ECU/t $CO_2$) | ExternE preliminary[2] (170 $/t C) (41 ECU/t $CO_2$) |
|---|---|---|
| mECU/pkm | | |
| Petrol car | 0.1 - 3.7 | 5.1 |
| Diesel car | 0.1 - 2.9 | 3.9 |
| Bus | 0.06 - 1.9 | 2.6 |
| Inter-city train | 0.1 - 2.9 | 4.0 |
| mECU/tkm | | |
| Heavy goods vehicle | 0.1 - 4.3 | 5.9 |
| Goods train (electric) | 0.1 - 4.0 | 5.5 |

[1] Estimates based on emission factors of German case studies.
[2] Preliminary results from the FUND Model, 'base case' assumptions, discount rate of 1%.

The uncertainty involved in the assessment of global warming impacts is very high. The range used in this study is based on the range of possible damage values reported by the IPCC of 5 to 125 US$ per tonne of carbon emitted. For practical reasons, the mean of this range was used for the damage estimates reported here. The full range of the results is illustrated in table 16.6, together with the damage resulting from applying preliminary results of the ExternE modelling activities on global warming.

A comparison with the results of a top-down study /Infras/IWW 1995/, which quantified external costs for the member countries of the European Union plus Switzerland and Norway, is presented in table 16.6. The different technologies and locations considered in the case studies lead to a considerable range for the damages. The lower bound of the results is in the same order of magnitude as the estimates reported in /Infras/IWW 1995/. But for road transport, the upper bound of the range is an order of magnitude higher. The results quantified for rail transport lie within the same order of magnitude for both approaches.

**Table 16.7:** Comparison of bottom-up and top-down approach (damages due to airborne pollutants and global warming)

| mECU/pkm or tkm | Bottom-up approach ExternE Transport | Top-down approach /Infras/IWW 1995/ |
|---|---|---|
| Car | 5 - 385[1] | 13.3 |
| Bus | 5 - 384[1] | 6.8 |
| Passenger train | 1.1 - 6.6[1] | 5.0 |
| Heavy goods vehicles | 44 - 312[1] | 23.6 |
| Goods train | 0.8 - 8.8[1] | 1.8 |

[1] depending on technology and location

Please note, that the externalities presented in this report only comprise the quantifiable damages due to airborne pollutants and greenhouse gases. Impacts due to accidents, noise, etc. which are expected to be considerable have to be added to these results.

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# EXTERNE

EXTERNALITIES
OF
ENERGY

Vol 9: FUEL CYCLES FOR EMERGING AND END-USE TECHNOLOGIES, TRANSPORT & WASTE

# PART III

# EXTERNALITIES OF WASTE

# EXTERNALITIES OF WASTE

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# 1. SELECTION OF TECHNOLOGIES, LOCATIONS AND IMPACT PATHWAYS

## 1.1 Scope of the study

Municipal solid wastes (MSW) are regarded as a major environmental issue in most industrialised countries as landfilling, the most traditional way of disposing of them, has become increasingly unacceptable due to lack of space and air and groundwater pollution. The main alternative that has been practised particularly in congested urban areas, incineration, has also been put under close scrutiny by environmental authorities due to its heavy air pollution burden. The European Union has issued several Directives defining policy criteria and emission standards to which waste disposal practices should adjust and member states have introduced appropriate measures into national legislation. In consequence the average cost of disposing of wastes by landfilling and by incineration have been sharply increased.

This has in turn already widened the scope for economic recovery of solid wastes. However, a common feeling especially within the most industrialised countries is that the availability of sites for solid waste landfilling and the social acceptability of incineration are rapidly vanishing due to their perceived adverse environmental impact.

On the other hand, solid waste landfills and incineration are regarded as an important alternative energy source. In Italy, it has been estimated that biogas generated by landfills and thermal or electrical energy that may be recovered from waste combustion represent a potential in the order of 2-3% of national energy requirements. Advantages of this energy source include independence from imports and some environmental aspects, notably net reductions of total atmospheric pollutants emissions with respect to what would result if wastes were incinerated and landfilled without any energy recovery.

The negative effects on the environment of MSW incineration should be explicitly compared with the above benefits. Environmental policy has indirectly taken such effects into account by adopting priority status for waste avoidance and materials recycling as well as by setting increasingly tight limits to emissions from new landfills and incineration facilities. As regards packaging, a major fraction of municipal solid wastes, the European Union (EU) in its Directive passed by the Council of Ministers in December 1993, has stipulated that landfilling should be limited within five years to only between 35% and 50% of the solid waste stream (depending on material), and that recycling should be privileged up to a target of 25-45% of total recovered packaging. In consequence incineration with energy recovery may account to up to 30-40% of total packaging output. Both recycling and thermal recovery should therefore be significantly increased with respect to current levels.

From an energy perspective, it is known that recycling is also usually beneficial up to a certain level. For some materials like glass and metals, recycling is also the only way to recover embodied energy. Gross energy savings from recycling of waste glass, plastics, and even more from paper, steel and aluminium, is estimated in the same order (per mass unit) of what could be obtained from incineration.

Let us remember, however, that there is only a limited trade-off between energy recovery from materials recycling and landfills or incinerators. Separation of incombustible matters like metals and glass prior to incineration or landfilling is the only way to recover energy that is embodied within these materials, with the possible exception of iron residues that could be magnetically extracted from combustion ashes. Separation of organic refuse (e.g. for composting) favours energy recovery through incineration as the loss of some combustible material is outweighed by the reduced moisture of the "fuel"; however, it reduces or even eliminates the production of energy-rich biogas if the refuse is landfilled. Biogas can be partly recovered, but a significant share is always dispersed into the atmosphere. Separation and recycling of plastics, paper and cardboard does reduce the calorific value of solid wastes and hence the potential for thermal energy recovery, and to a lesser extent also the generation of biogas that may be obtained if the same refuse were landfilled.

On the other hand recycling processes also display some environmental and other external effects that should be compared with those of avoided processes for production from raw materials. Furthermore, the second law of thermodynamics entails that recycling would actually require higher energy (and possibly also raw material) inputs, so that beyond a certain level its desirability vanishes on both economic and environmental grounds. Net energy which may be recovered through recycling, after allowing for collection, sorting and cleanup, may be much less than the theoretical energy content of materials, and in some cases could even be negative, as stressed by Pearce and Turner (1992).

The cited E.C. Directive underlines that suggested targets are subject to revision as more research and experience emerges concerning their feasibility. Indeed, targets set by the Commission as well as by National Governments for both recycling and thermal recovery can be regarded as largely political decisions that do not derive from evaluation of relative costs and benefits, and notably do not explicitly account for externalities.

The evaluation of external costs arising from several technical options that involve municipal solid waste recycling or other forms of energy recovery from them is therefore valuable for both environmental (solid waste) policy and for energy policy analysis.

Among industrial processes currently run in the field, we may recall the following:

(1) biogas recovery from MSW landfills;
(2) mass burning (incineration) of MSW;
(3) production and burning of residual derived fuel (RDF), possibly with simultaneous production of compost;
(4) recycling of wastepaper and cardboard;

(5) recycling of glass bottles;
(6) recycling of aluminium from scrap vehicles, beverage cans and other food containers;
(7) recycling of tinplate and other iron residuals;
(8) recycling of plastics.

The current project only considers options (1) and (2), that is energy recovery by mass burning of MSW and by collection and utilisation of landfill biogas. However, this choice does not entail any judgement about the energy saving potential or the overall desirability of other waste treatment options, and their likely external effects compared to options not considered.

The main reason for this choice is that these options have the largest energy potential *likely to be recovered in the medium term* (five to ten years) in the EU, consistent with policy options chosen by the EU as well as by National Governments; another good reason is that incineration and landfilling represent broader range options for MSW disposal; moreover, as stated above, they would still be required to a significant extent even if the highest recycling targets now foreseen were achieved.

Finally, it ought to be noted at this point that, while a huge amount of scientific research has been devoted to almost every aspect of the environmental impact of MSW incinerators and landfills, no comprehensive studies existed until very recent times, to our knowledge, attempting to evaluate their total external costs. An exception is given by the study prepared for the UKDOE (1993). The present analysis differs from that as it considers incineration and biogas recovery as energy saving options, and looks at the externalities from the latter as an avoided cost (i.e. a benefit) of the incineration process, computing a final *net* external impact of the fuel cycle defined in this way. Further, our study explores more deeply aspects that are crucial for our perspectives, such as the "perceived" impact from the landfill, estimated as a loss of property values in the neighbourhood of the plant. Finally, the methodology of the present work is consistent with that developed within the *ExternE* accounting framework, while the UKDOE study is mainly based on previous research by CSERGE (1992) regarding social costs of energy cycles.

## 1.2 Impact analysis of MSW-related cycles: some methodological issues

### 1.2.1 The waste disposal constraint and its consequences

Energy production by solid waste incineration or from landfill biogas may well be regarded as a fuel cycle. Indeed, much of the external impact closely resembles that of fossil fuels, although emissions differ quantitatively and some of the most important impacts are more specific to waste disposal. As a consequence, it will be possible to some extent to exploit and adapt the same methodologies and tools that were developed within the ExternE framework for other fuel cycles (notably coal, EC DG XII 1995c). However, some impacts are peculiar to the MSW cycles considered and will deserve special attention within the study.

A special feature of these cycles that must be recalled derives however from the basic fact that *solid wastes must be always disposed of in some way*. Thus, the evaluation of external burdens and their impacts must always be compared with a benchmark alternative so that this "mass constraint" concerning waste disposal is respected.

As a consequence of this constraint, any technology leading to energy recovery from wastes involves important external *benefits* as well as costs that would derive from the displaced alternative way of disposing of MSW.

Both the energy recovery technology and the benchmark alternative could be specified in several ways. Let us outline three sets of assumptions that seem worth considering:

I. Historically, the traditional way of disposing of MSW has always been landfill dumping (without energy recovery). Therefore, one could consider either (IA) incineration or (IB) biogas recovery as a way to recover energy and assess their impacts in turn, taking the "old dump" as the benchmark.

II. Lately, legislation in most EU countries mandates biogas recovery from MSW landfills for environmental and safety reasons. Biogas is then either utilised for energy (usually electricity) production or flared. Hence, one could consider either (IIA) incineration or (IIB) biogas utilisation as a fuel cycle to be assessed against the benchmark of MSW landfilling with biogas collection and flaring.

III. Finally, we may just consider one cycle, that is energy production by MSW incineration, and regard the landfill with biogas recovery *and* utilisation as the benchmark alternative: this could still be considered as a remarkable energy saving option as a tonne of MSW incinerated would yield between 300 and 900 kWh, depending on MSW composition and conversion efficiency, while biogas recovered from one tonne of waste is likely to yield only 40-70 kWh.

Note that the first approach is more consistent with the methodology adopted within the ExternE programme, which does not usually take any existing legislation for granted but evaluates impacts, so that results could be in principle used for the assessment of any environmental policy. Since many landfills within the EC are not yet equipped with effective biogas recovery (According to De Poli and Pasqualini (1991) in Italy, there are some 2000 landfill sites, of which 480 are authorised, 73 equipped with biogas collection, and only 12 with energy utilisation; this means that less than 1% of LF biogas energy potential is currently exploited) this approach is also highly significant if we are interested in the change in environmental impacts that could be entailed by new incineration plants and landfill sites equipped with energy recovery.

However, the second and third approaches are more relevant as to the choice faced by decision makers concerning new plants that will be all equipped with biogas collection and (possibly) utilisation.

The present study begins by assuming approach (III) defined above, which is quite representative of European MSW policy issues, if not always of the actual disposal status. A benchmark consisting of a landfill site without biogas recovery amounts to considering a solution which is out-of date and ruled out by modern regulation and would involve analysis of some peculiar impact pathways, e.g. horizontal migration of biogas and the related blast dangers, which is not easily carried out.

The "fuel cycle" considered will therefore consist of several stages that are listed below; each stage involves externalities that will be denoted by symbols in brackets:

- a lorry transporting MSW from town to the incinerator (TRASPINC);
- an incineration plant with electricity production (INC);
- a lorry transporting incinerator residuals to a landfill (TRASPASH);
- a landfill site for incineration residuals (ASH).

These combined technologies allow society to avoid the (external effects of a) landfill site (LFL), where:

- wastes are transported by lorry (TRASPLND);
- landfill biogas is collected and burned, with energy recovery (COL);
- part of the gas cannot however be collected, hence it is dispersed (DIS).

Few remarks need to be added as regards the *waste transport* stage. Normally, the externalities occurring during the fuel production and transport stages are added to the overall external cost of the fuel cycle (e.g. as in the coal cycle). In this case, however, it clearly appears that wastes are *not* a normal fuel, produced and transported to the treatment plant *with the only aim* of producing energy. It is rather an undesirable residue that for hygienic reasons must be transported somewhere *away from* the place where it has been created, in any proper disposal facility. It would thus be improper to consider externalities from transport simply as belonging to the fuel cycle (III) sketched out above. Moreover, should we include transport in the fuel cycle considered, it would then become difficult to identify a clear criterion for the inclusion or exclusion of other stages related to the waste's life cycle: for instance, why waste transport and not waste collection? Therefore, in principle, we hold that the analysis of energy recovery from waste should *not* include the transport stage in the fuel cycle assessed here. On the other hand, if the analysis aims at comparing MSW incineration and landfilling, it has to be noticed that there usually arise a *differential* impact in terms of transport cost which cannot be neglected. In fact, landfills are usually located further away from urban areas (the origin of wastes) than incinerators, since they require larger areas and are even more strongly opposed than incinerators by local communities. Thus, our assessment of the *net* impact of the incinerator will include the transport external cost, but only in terms of the *differential* impact of waste transport to the incinerator *versus* the alternative transport of the same quantity of wastes to the benchmark plant examined.

For reasons discussed below in detail (Section 1.4), it is reasonable to assume that :

- impacts from energy recovery of landfill gas should not dramatically differ from those of flaring if measured as total impacts;

- impacts of a landfill site for MSW are not very different from those of a site for incineration residuals, with the exception of biogas production which is either burned or dispersed.

Based on all the hypotheses previously specified, we may express the total external impact of landfilling relevant to the comparison made here as:

$$L(W) = TRASPLND(W) + COL(W) + DIS(W) \quad (1)$$

while the total impact of waste incineration is given by:

$$I(W) = TRASPINC(W) + INC(W) + TRASPASH(W) + ASH(W) \quad (2)$$

where W is the amount of MSW treated, from which an electricity output X(W) is recovered in the incineration plant, while an electricity output Y(W) is foregone, which might have been generated by the biogas recovery process in the landfill disposal.

The *net* externalities entailed by approach (III), in terms of ECUs per tonne disposed, would therefore be:

$$I(W) - L(W) = [TRASPINC(W) + TRASPASH(W) - TRASPLND(W)] + [INC(W) + ASH(W) - COL(W) - DIS(W)] \quad (3)$$

In terms of external costs per net kWh recovered, correspondingly, we would have:

$$EXT(X-Y) = [TRASPINC(X) + TRASPASH(X) - TRASPLND(X)] + [INC(X) + ASH(X) - COL(Y) - DIS(Y)] \quad (4).$$

In other words, consideration of the constraint that MSW must be disposed of in some way entails very important consequences for the selection of priority impact pathways to be analysed and consequently for the final results given.

As mentioned before and shown by equations (3) and (4), this study will be focused on *differential* externalities arising from treatment of MSW that leads to energy recovery. Other externalities that may be quite relevant in general but are not likely affected by resorting to (e.g.) incineration rather than landfilling, will therefore be mentioned but not examined in detail.

The study will evaluate externalities both in terms of equations (3) (i.e. in terms of cost per waste mass disposed of) and (4) (in terms of cost per kWh recovered). Since we are interested in incineration and biogas recovery as an energy saving process, more emphasis will however be put on the final results in terms of mECU/kWh. In order to reach estimates of (3) and (4), the study will obviously have first to find results in terms of equations (1) and (2).

A careful accounting rearrangement of externality items and energy production rates arising from different stages should allow the reader to evaluate other approaches as defined above. Under the assumption that externalities of biogas flaring are close to those of biogas burning with energy recovery, i.e.

$$COL(Y) = COL(0)$$

and

$$DIS(Y) = DIS(0)$$

we might indeed consider approach (IIA) and evaluate:

$$EXT(X) = INC(X) - COL(0) - DIS(0) \quad (5)$$

while externalities of approach (IIB) would be negligible.

Finally, the evaluation of externalities of dispersed biogas under the no collection assumption would be the basis of possible future work leading to the evaluation of approaches (IA) and (IB) which are not considered further in this study.

### 1.2.2 Impact classes and valuation methodology

Two kinds of *burdens* are analysed: those assessed by scientific literature, and those perceived by the community. In principle, these different definitions of damage correspond to different techniques of monetary evaluation. The first is basically the dose-response approach (DRA) (sometimes also called Exposure-Response Approach) as described e.g. in ORNL (1991); or as adopted in the work of O. Hohmeyer (1988), later adopted in the ExternE accounting framework. It involves as main steps: (i) identification of burdens of the impact of interested technologies; (ii) identification of impact categories; (iii) outline of pathways for each impact; (iv) evaluation of dose-response mappings for impact pathways; (v) estimation of external costs related to each impact category.

The second approach is based on *direct* evaluation of willingness to pay (WTP) by affected people in order to avoid the impact of technologies. In turn, this evaluation can be performed by several techniques which are well known by cost-benefit analysts and environmental economists and are reviewed e.g. by Braden and Kolstad (1992) or by Cropper and Oates (1992). The most useful techniques for industrial plants investigated in this study are hedonic pricing of land and estates, and contingent valuation.

These approaches can lead to very different results in estimation of the same impacts because often local communities perceive damages differently to 'experts'.

The choice of a "holistic" approach that privileges the assessment of the overall perception of the facility's impact rather than detailed "reductionist" analysis of several impact pathways is

not however a technical one. Psychologists as well as economists have suggested several reasons that may explain sharp differences between lay and expert assessment of environmental impacts and other risks, so that the choice is to a large extent philosophical. Nevertheless, in the case of technologies for energy recovery from MSW such as biogas utilisation, our working hypothesis is that there should be no large differences between scientifically assessed and perceived impact, while larger differences may occur in the case of incineration plants and landfill sites. In the last two cases, perceived impact may be only partly related to the amount of recovered energy but to the very existence of the plant itself. However, there is no reason to exclude that size of the plant also plays an important role in impact perception, as UKDOE (1993) seems to imply.

In this study, we will follow the impact pathways approach as adopted by the ExternE framework for all technologies, but impacts perceived by local communities will also be described and assessed for the landfill site using an hedonic pricing technique. Most of the impact pathways hereby evaluated with the DRF approach will be similar to those assessed in more "traditional" (fossil) fuel cycles. Some of them, however – notably those concerning trace organic compounds and heavy metals arising from landfill sites and incinerators - will involve the definition of new exposure-response functions and impact pathways.

The following sections review the main phases of the reference technology and their burdens.

## 1.3 Waste incineration: technology outline and review of environmental impacts

### 1.3.1 Technology outline

Waste incineration is the most used waste disposal technology after landfilling. Increasing use of this solution in the last decades, notably in densely populated areas of most developed countries is mainly due to a few remarkable advantages. Firstly, it allows a larger energy recovery. Energy can be recovered either as steam which can be used as thermal energy for district heating, or as electrical energy. Assuming a low heat value of 8,000-16,000 kJ for kg of waste, it is possible to obtain about 0.3 – 0.9 kWh for each kg of incinerated matter, depending on thermal efficiency. Secondly, it requires a much reduced use of space. While bottom ashes and other solid incinerator residuals must be landfilled anyway, these represent typically only 30% of the original weight and 10% of the original volume of the waste that has to be disposed of.

Waste incineration in Italy covers about 6% of total waste disposal. Larger percentages can be found in other European countries (see Table 1.2).

There are several modern MSW incineration technologies that meet emission standards required by EC and national legislation. The incinerator considered in this study is a new plant which has not yet been completed, although approval has been given. It will be located in the western part of the outskirts of Milan, about 5 km far from the storage centre where waste

collected in the town are provisionally placed before their transport to the treatment plants. Our analysis has been based on the emissions data provided by AMSA (the local public utility responsible for waste collection and treatment) concerning the future performance of the plant (more details on emissions and technology data are given in Appendix 1).

The planned plant consists of a grate-type furnace, with secondary combustion chamber and energy recovery. It has a selective non-catalytic reduction (SNCR) emission control system with ammonia injection in secondary combustion chamber and dry-type tail-end system with final baghouse filters for removal of acid gases, particulate matters and toxic micropollutants through addiction of hydrated lime and activated carbon.

The waste treated comes from the city of Milan, where separate collection for glass, paper, plastic, cans and organic matter has been introduced for about one year. The average waste calorific power - after separate collection - is around 15000 kJ/kg. More details on the average waste composition in the urban area of Milan are given below and in Appendix 1.

The capacity of the specific plant analysed is about 900 tonnes/day, working for 7440 hours/year (85% utilisation rate), with a gross electricity output of 247.7 Gwh/year.

For consistency with features of the corresponding landfill site, the plant is assumed to be working for ten years at the given rate. The evaluation of external effects from the incinerator, apart from the occupational accidents impact, is not affected by this assumption.

Further characteristics of the plant affecting atmospheric emissions and their dispersions will be discussed in section 2.1.

### 1.3.2 Scientifically assessed environmental impacts

Municipal waste combustion emissions have been recently studied by many scientists, in particular their health effects. However, despite the big research effort in this field, many impacts have still to be quantified, or perhaps even discovered.

Pollution created by incineration processes is largely atmospheric. Toxic substances entering the incineration plants leave it as flue gases, bottom ashes and slag, dust trapped by air pollution controls, and fly ashes, liquid waste streams are typically of little significance. The mass balance among these kinds of output for each particular pollutant depends on its specific chemical and physical characteristics, among which the melting point is usually crucial. Mercury is the most volatile metal (see Table 1.1).

Bottom ashes, slag and dust trapped by filters are transported to landfills and disposed of there, therefore (assuming similar degree of leakage) they are not likely to cause a relevant *differential* impact with respect to this waste treatment technology, except for the dust which could be windblown during transportation. Analysis of these leakage's is not practical as releases occurring in transportation can hardly be mapped into burdens. Atmospheric

concentration and soil deposition increases depending on them should however be very small compared to those deriving from flue gases, so that this pathway will not be further analysed.

**Table 1.1** Partitioning of some metals by MSW incineration (Source: Brunner-Mönch, 1986)

| | **Flue gas** | **Electrostatic precipitator dust** | **Slag** |
|---|---|---|---|
| Cadmium | 12% | 76% | 12% |
| Copper | 1% | 0% | 89% |
| Iron | 0.02% | 1% | 99% |
| Lead | 5% | 37% | 58% |
| Mercury | 72% | 24% | 4% |
| Zinc | 4% | 45% | 51% |

Impacts deriving from landfilling bottom ashes and exhausted air pollution control devices like electrostatic precipitators are those indicated in section 1.2.1 above as ASH. Their impact is qualitatively close to that discussed for a MSW landfill (see below), although different characteristics of landfilled material could determine differences in their environmental effects. In particular, metals contained in bottom ashes may be more volatile than those contained in landfilled MSW and therefore more likely to contaminate individuals directly or through the food chain (Brunner-Mönch, 1986). On the other hand, their reduced volume allows (and calls) for more careful manipulation and stricter containment techniques in the landfill. Likewise, leachate that may leak from the landfill would be much less by volume but more "concentrated". Little more can be said of this comparison in spite of widespread scientific research about the environmental impact of landfill sites.

Atmospheric emissions, on the other hand, may well directly damage the environment surrounding the incineration plant. Usually these substances are divided into micro- and macropollutants.

*Macropollutants* are those substances generally produced in large quantities: Total Suspended Particulates ($PM_{10}$), sulphur oxides ($SO_x$), carbon oxides ($CO_x$) and nitrogen oxides ($NO_x$); in our case we may also consider hydrogen chloride (HCl) and fluoride (HF).

The quantities of macropollutants emitted by incinerators are of course worth considering for their impact on the environment. However, it should be stressed that in terms of *absolute* values - instead of values per unit of energy produced - these are small quantities compared to those annually emitted by any coal or oil plant for electrical energy production.

**Table 1.2** Weight (Mt) of MSW treatment options in some countries (*)

| Country | Year | Total | Landfill | Incineration | *of which: % with energy recovery* | Composting | Recycling |
|---|---|---|---|---|---|---|---|
| Canada | 1992 | 18800 | 14070 | 1101 | *8.30* | 218 | 3400 |
| USA | 1993 | 187790 | 117028 | 29933 | *7.60* | 5821 | 34929 |
| Japan | 1991 | 50767 | 19379 | 3616 | *NA* | 57 | 1688 |
| Austria | 1990 | 2506 | 1700 | 310 | *100.00* | 85 | 400 |
| Denmark | 1993 | 2377 | 468 | 1500 | *100.00* | 206 | 203 |
| Finland | 1990 | 3100 | 2400 | 50 | *100.00* | 50 | 600 |
| France | 1992 | 20500 | 9500 | 7600 | *72.40* | 1300 | 740 |
| W.Germany | 1990 | 21615 | 14219 | 6039 | *NA* | 369 | 0 |
| Italy (**) | 1994 | 26910 | 23675 | 1372 | *49.00* | NA | 1828 |
| Netherlands | 1991 | 7602 | 3610 | 2500 | *93.00* | 475 | 578 |
| Spain | 1993 | 14256 | 12061 | 1560 | *78.00* | 1560 | 0 |
| Sweden | 1990 | 3200 | 1400 | 1300 | *NA* | 100 | 400 |
| UK | 1989 | 20000 | 14000 | 2500 | *50.00* | NA | 1000 |

Source: OECD Environmental data, 1995;

(*) The different treatment methods are *not* mutually exclusive. Incineration and landfilling are a typical example, since solid residuals of the former have to be disposed of again by the latter.

(**) Source: Our elaboration on Italian Ministry of the Environment, *Relazione sullo stato dell'ambiente*, 1997

Much less considered in previous studies, compared to the other macropollutants, was hydrogen chloride (HCl). In our specific case this pollutant is emitted in considerable quantities *per kWh*, mainly due to the presence of plastic materials in solid wastes incinerated. HCl can be a notable source of acidic deposition and materials corrosion. If its presence in the work environment is large enough, it may also damage incinerator workers, causing eye and throat irritation at least.

Hydrogen fluoride (HF) is likely to cause similar damages. However, in this case the quantities emitted are likely to be of small concern.

A *micropollutant* is any substance emitted in very small quantities, usually just in traces, but which in spite of this may have adverse effects on human health and the environment. Most recent studies on waste combustion plants focus on the impact of specific micropollutants, the effects of which are not yet well known (Radian Corp., 1989).

They can be divided, first of all, into *organic* and *inorganic* substances.

Within the first group, Dioxins (polyChlorinated-Dibenzo-Dioxins, pCDD) and Furans (polyChlorinated-Dibenzo-Furans, pCDF) are regarded as most dangerous for the environment and human health. Both pCDD and pCDF will be refered to as "dioxins" in the following pages, and sometimes measured as 2,3,7,8-Tetra-Chlorinated-Dibenzo-Dioxins equivalent (TCDDe). They have been studied for years by many scientists, trying to deepen scientific knowledge about their chemical properties and their toxicity for human health. Due to some past tragic events (e.g. the 1976 accident at a chemical plant in Seveso), they are also strongly opposed in the collective perception of environmental risk, and severe standards have been established in most EC countries. Unfortunately, there is still scientific uncertainty, even as regards the process that generates them (Commoner et al., 1987).

Beside dioxins and furans, some studies (Radian Corp., 1989) tried to evaluate other organic pollutants, which are thought to be dangerous for human health: Polycyclic Aromatic Hydrocarbons (PAH), PolyChlorinated Biphenyls (PCB) and Formaldehyde (HCHO).
Some problems arise, however, in attempting to evaluate the impact pathways of those pollutants. In fact, it is quite difficult to find reliable representative data on their emissions. They are rarely measured, owing to the fact that there is no specific standard for them. Sometimes they are grouped and measured with many other different organic pollutants, but it is not feasible to establish with an even rough degree of approximation which portion of the quantity measured can be attributed to the specific pollutant we are considering.

Anyway, since their toxicity is thought to be much less than that of pCDD-pCDF, we will restrict our analysis to these two organic pollutants, without relevant losses in the accuracy of the evaluation. As it emerges from the meta-analysis carried out by A. Levin et al. (1991), comparing incinerator health risk assessments performed in 21 different plants in USA, the greatest contribution to health (cancer) risk is clearly attributed to pCDD-pCDF and cadmium emissions.

Concerning inorganic micropollutants, incinerators emit small quantities of different heavy metals. As, Be, Cd, Cr, Hg, Pb, are those which can be found looking through the specific literature. As mentioned before, their presence in flue gas or in bottom ashes depends heavily, for a given composition of municipal solid wastes, on their physical and chemical characteristics.

Among metals reported, cadmium (Cd), lead (Pb) and mercury (Hg) are extensively analysed in environmental risk assessments, owing to the quantities emitted and/or to their specific toxicity factor.

As noted above, both organic and inorganic micropollutants are potentially dangerous sources of damage for vegetation, animals, and human health. A strong negative characteristic they show is that of being *persistent*, i.e., they cannot be reduced to an harmless state by natural agents (heavy metals) or they can be reduced but only after several decades (dioxins). Therefore, not only can they accumulate in the environment, but they are also transported and either dispersed or accumulated (by atmospheric agents, or through the food chain) in some unpredictable ways, giving rise to potential dangers also in the long run.

### 1.3.3 Other impacts perceived in the neighbourhood of plants

As often happens, the community living near a plant that is a source of pollution perceives some effects which can be different from those scientifically assessed, or which can hardly be estimated through a scientific analysis of impact pathways.

Odours coming from the plant, traffic transporting wastes and residuals of combustion, or visual impact are typical environmental damages which cannot be evaluated only by a dose response analysis (except for some specific effects like e.g. the reduced visibility caused by high concentration of particulate matter in the air, which is however a minor effect in this case). Its evaluation requires therefore the use of direct estimates of the consumer's WTP to avoid these effects (or to get rid of the plant), that is contingent valuation or (preferably) hedonic pricing as these impacts are likely to affect prices of housing in neighbouring areas.

## 1.4 Landfill sites for MSW: technology outline and review of environmental impacts

### 1.4.1 Technology outline

Landfills are usually natural or artificial holes, hollows or subsidences devoted to the dumping of municipal or other (possibly treated) solid wastes for permanent storage. In modern plant for MSW which we will be considering in this study, wastes are isolated from the underlying soil by means of highly impermeable matters (e.g. clay) and/or artificial liners. Refuse is compacted and covered by more impermeable strata at the end of the cultivation period in order to favour some reuse of land. Rainwater penetrating inside the landfill or moisture of wastes dripping out of it (leachate) has to be drained and treated to avoid soil and groundwater pollution.

In addition to this, micro-organisms fed by organic substances in MSW tend to produce, at changing rates over time (see section 2.2), biogas made of methane, carbon dioxide, nitrogen and several other gases in traces. This biogas can be collected by wells laid in the landfill.

Landfilling is the main technology adopted for MSW disposal in most countries. Leaving aside external (usually environmental) costs, it is the cheapest option even if the best available technologies for waste containment are used, and this explains most of its success. On the contrary, energetic recovery of biogas produced is less frequent. A 1991 world survey (Willumsen and Bach) listed 249 plants in the world with an energy production of 1.1 million tonnes, about 70% of which occurred in the US and 25% in the E.C.. The lack of biogas exploitation is partly due to the fact that many "old fashioned" landfills produced little methane due to prevailing aerobic conditions and were not suitable for biogas collection. Atmospheric emissions of such facilities were usually of the same kind of modern plants except for reduced methane generation; little could be done to curb them.

On the other hand, anaerobic conditions prevailing in modern landfills lead to the production of a biogas with an important energetic value as methane content averages nearly 55%, while carbon dioxide is still the second main component (40%). The remaining part is composed of highly variable shares of hydrogen, oxygen and especially nitrogen and water vapour, plus a hundred trace gases that are of large relevance in the characterisation of atmospheric impacts (though not necessarily involving large external costs).

Biogas collection is often prescribed by landfill regulation as a means of preventing methane blastoffs and subsequent fires. Otherwise, biogas recovery and exploitation would be economical only for large plants. Indeed, even if a collection grid exists biogas is often flared in most cases.

The capacity of the landfill site analysed is more than twice as high as that of the incineration plant, reaching about 2000 tonnes/day for 330 days/year. The average duration of a landfill site is 10 cultivation years: the required volume of the site for such a disposing capacity is about 20 million cubic meters, depending on compaction and waste characteristics. After the site is closed biogas generation, collection and recovery is assumed to continue for 14 *more* years. The site of Cerro Maggiore, considered for this study, has been closed very recently, under the pressure of a heavy local opposition. As it was not possible to obtain a *complete* set of data on this site, some of the data used in the following analysis do not represent the "actual" characteristics of the plant analysed; they are anyway still a reasonable and plausible estimate of them.

Unlike the real site, the landfill site considered in this study is assumed to be equipped with energy recovery of collected gas. Collection efficiency is assumed to be 60%. Electricity is produced by four Otto-cycle four stroke engines with a total nominal capacity of 4.0 MWe and gross energetic yield of 12.3%. Technical features of flue gas generation are given in section 2.2 as they affect atmospheric emission and their dispersion.

These characteristics entail a total electricity generation of about 720 GWh over 24 years (though at a variable annual rate). This figure also corresponds to recovering an average 48 kWh per tonne of landfilled MSW, i.e. 5.5% of what may be obtained from MSW incineration.

### 1.4.2 Scientifically assessed environmental effects

The environmental impact of a MSW landfill can be classified according to several dimensions.
"Routine" impacts ought to be distinguished from what may arise in case of accidents.

As regard receptors, the taxonomy includes air impacts of biogas emissions, soil and groundwater impacts of leachate, and other effects less prone to scientific analysis but highly perceived as being serious by communities living in the surroundings of the plant.

As we discussed above, environmental impact of a landfill depends on MSW characteristics, landfill design, geology of the site, isolation systems, and provision for biogas collection and its utilisation. Atmospheric emissions of normal operation of a MSW landfill worth considering include:

- carbon dioxide ($CO_2$);
- methane, as not all production can be collected but some is released particularly from landfill borders;
- steam, whose amount is probably neither large nor worth considering.

The above and nitrogen make up over 99% of biogas, though in varying proportion according to MSW composition, density and age of the landfill. Much concern can however be attributed to trace gases, which have been listed in large numbers (Rettenberger and Stegmann, 1991):

- hydrogen sulphide ($H_2S$);
- sulphur-organic compounds, notably mercaptans;
- chlorine-organic compounds;
- aromatics.

If biogas is partly recovered and either flared or used for energy production, the emissions from the flare or combustors should be considered. They include all of the former flue gases but in a quite different proportion. Among macropollutants, methane is largely replaced by more $CO_2$ and by $NO_x$ emissions; $H_2S$ and other sulphur-organic compounds are turned into limited amounts of $SO_2$.

As for micropollutants, complex and hardly known changes in the chemical and physical characteristics of trace emissions may occur. However, permanence of flue gases at high temperature during the combustion process favours elimination of most of them, so that this pathway may be reasonably neglected.

Soil and groundwater pollution under normal operation derive from leakages, which can usually be regarded as negligible, and from treatment and disposal of collected leachate. The latter will increase BOD/COD values if disposed into water streams; however, if it is conveyed to a sewage treatment plant, its external cost is largely internalised. The internalisation is probably not complete, as there are always some environmental impacts left - e.g. those arising from sludge disposal, or dispersion into the environment across various stages of the "leachate cycle" - but these can be regarded as minor.

Accidental impacts of landfills may be more serious than those of normal operations. Two main classes of accidents should be considered:

1) Ruptures in the containment system of the landfill, due to dumping of hazardous materials not expected in the planning stage and therefore not suitable for a MSW landfill site, breaks of protective liners, landslides of the landfill walls, excessive roof pressure, failures of the leachate drainage system, mistaken evaluation of several geological features of the site like perched water. These may lead to more substantial leachate spills and widespread pollution of soil and groundwater.

2) Fires (usually originated by methane blastoffs).

In the former case, it is difficult to assess both size and probability of adverse events. Partial evaluation could be tried by examining averting behaviour costs in case of some major accidents, e.g. the cost of alternative water supplies and that arising from disease outbreaks. This approach seems more promising than looking at separate effects of pollutants which are of the same kind of leachate but are dispersed to a much larger extent into the environment.

As regards fires, some of the air pollutants that would arise are the same as normal emissions, but very different concentrations strikingly alter impact pathways. Furthermore, the role of some trace gases like dioxins and furans in such events is likely to gain an important role in the assessment of these events. However, modern plants equipped with biogas collection display very low blast-off probabilities, so that this impact will not be considered for scenarios envisaged in section 1.2.1 for the current study.

### 1.4.3 Other impacts perceived in the neighbourhood of plants

As in the case of incineration (and possibly even more) impacts perceived by the community living in the surroundings of an active landfill are certainly not limited to air and water pollution. Some impacts cannot be quantified using the dose-response approach:

- Odours depend on several meteorological factors and are the joint product of many trace gases; it seems more sensible to proceed to direct evaluation of behavioural or stated nuisance as this cannot be attributed to the individual role of each trace gas. However, since dispersion of the main malodorous compounds can be analysed as for any other gas, attempts will be made to "measure" expected odours arising from the plant. Such measures will later be used as determinants of WTP values to get rid of the landfill and may be useful to ensure higher transferability of results;
- Landscape damages also require a direct WTP estimation; it may be noted that these can be partially offset by the "land recovery" effect as hollows and old quarries may be returned to some uses so that landscape improvements are attained some time after the end of cultivation;
- Traffic congestion, increased emissions, and waste and odour losses of trucks heading for the plant may not be seen as a by-product of the landfill as an energy facility, because they

are found in all technologies dealing with waste disposal or recycling. Evaluation of some components (local traffic congestion and the increase in traffic emissions due to the congestion effect) of the related external costs by dose-response approach is possible in principle but likely not to hit the real points of people's complaints about such "traffic". Again, WTP direct estimation and hedonic technique seem more suitable, if not precise, for evaluation of these impact factors.

Finally, operating landfills are a favourite breeding area for birds, mice and other animals which are carriers of pathogenic micro-organisms commonly found among MSW, while the problem would be much less serious with landfills containing only bottom ashes and inorganic materials. There is little or no data on the epidemiology around landfill sites, so that it is difficult to quantify this effect according to the impact pathway methodology. We hold that this impact is not likely to change significantly the quality of the results presented here.

## 1.5 Selection of priority impacts

### 1.5.1 Overview of burdens and impact pathways and selection criteria

Identified burdens of MSW incineration and landfilling, their impacts and likely relevance are summarised in Table 1.3 and regrouped in order to discuss selection of priorities. The Table identifies the immediate receptors of burdens so that they are split according to traditional classification, and actual targets on which impacts must be evaluated.

In the previous sections, presentation of impact pathways has been followed by discussion of their likely relevance, which is summarised in the last column of the table. In general, all impact pathways marked as medium or high impacts should be analysed.

The actual relevance of impacts is affected by the features of technologies and locations chosen for the study, with related pollution dispersion patterns. Moreover, since impacts of incineration and landfilling of its residuals often displace those of a MSW landfill site, some important pathways could be neglected because their *net* impact is minor, or difficult to assess in differential terms.

The selection of priority impacts that will be further analysed in this study depends on several factors, among them mainly the availability of data, dispersion models, dose-response functions (DRF), and values to be attributed to impacts. Some impacts could be evaluated only partially; expected ranges of impacts not properly estimated will be supplied by means of simpler techniques, e.g. by a critical transfer of impacts and costs estimated for similar cycles on a burden/output basis.

In order to understand more precisely issues of data, DRF and value availability for this fuel cycle, it is useful to distinguish between common and specific impacts. This classification partly overlaps with that between macro- and micropollutants.

Common impacts arising from common pollutants, are mostly regional or global, sometimes also local but of standardised character. We define them "common" as they can be easily compared with those of other (notably fossil) fuel cycles. This applies to:

- atmospheric emissions of carbon dioxide and methane which are mainly feared for their contribution to the greenhouse effect;
- sulphur and nitrogen compounds which may be considered for their health effects as well as for their contribution to acid depositions;
- particulate matters ($PM_{10}$) dispersed into the atmosphere to the extent that their composition is not defined in higher detail;
- chlorine compounds which may contribute to ozone layer depletion alongside CFCs;
- local and regional impacts of traffic.

For all of these components, effects should be evaluated strictly in line with other fuel cycles as analysed within the ExternE accounting framework.
Main specific impacts concern:

- health effects for general populations of trace gases released both during normal operations and in case of fires. The most feared such gases are aromatics, dioxins and furans. Effects depend on atmospheric dispersion of pollutants and population density in the affected area;
- damages to groundwater: these may arise mainly from spills from the landfill for accidental reasons, e.g. ruptures of isolation liners. Their effects are hardly evaluated as no satisfactory studies exist about frequencies of such adverse events;
- visual impact and annoyance arising from odours and local traffic of waste trucks that are highly specific and hardly evaluated by the impact pathways approach so that direct evaluation of monetary damages should be implemented.

### 1.5.2 Priority impacts of incineration

As regards the environmental impact of incineration, *human health* is no doubt the most important subject of scientific studies as well as the key class of impacts worth evaluating by means of the dose-response approach, for several reasons.

Firstly, the incineration plants are usually located - as in our case - near (or inside) an urban area, so that the population density in the surrounding area (which is more exposed to atmospheric pollution) is relatively high.

Secondly, due to the peculiarity of pollutants emitted, the main *known* effects are those referred to human health. In fact, since scientific studies on the effects of these micropollutants began quite recently, it is not easy to find reliable dose-response functions referred to the impact on other possible receptors which obviously receive less consideration than man, like cattle or poplar trees, even if they are likely to suffer some negative impacts.

Among human health impacts, let us first of all distinguish between *direct* and *indirect* pathways. ***Direct*** impact pathways are those followed by air pollutants damaging human health directly through air inhalation.

***Indirect*** impact pathways will be, on the other hand, those occurring after pollutant deposition on soil or water surface. In this case, man can be exposed to these pollutants by ingestion of contaminated food or water, or of contaminated soil risen up by the wind and dermal contact - mainly by children.

The direct impact pathways have been studied and quantified - to the best accuracy possible, accepting the existence of uncertainty - not only for macropollutants generated by traditional power plants, but also for some micropollutants emitted by waste incineration processes.

On the contrary, the indirect pathways still need further and deeper analysis before they can reach some *quantitative* estimate of risk. These estimates require firstly a deep - and difficult - analysis of the quantity of each pollutant which can be taken up in different vegetal and animal tissue without any apparent evidence - i.e., without any kind of pathology which could in some way show their presence in tissues and thereby prevent people from eating them. Then they also need a reliable description of the average diet of the people living in the places where these contaminated vegetal and animals are consumed. Finally, it has to be estimated the specific dose-response function linking the quantity of pollutant ingested to the possible pathology of the human body.

As this chain of effects has a very high degree of complexity, we chose to describe indirect effects only qualitatively. In fact, models for quantification do exist, but data requirements are very demanding and results rather approximate. (For a review, see Radian Corp. 1989, chapters 3, 29-56; Villars et al. 1992). In order to give a rough quantitative idea of the order of magnitude of the effects involved it must be noted that according to some recent studies (Levin et al., 1991; Cernuschi and Giugliano, 1991) *indirect effects of micropollutants on human health should be at least as relevant as the direct one.*

**Table 1.3a.** Summary of expected burdens and impact pathways

| Burden | Pollution vehicle | Target | Impact | Expected relevance and difficulties |
|---|---|---|---|---|
| **Incineration (INC)** | | | | |
| $PM_{10}$ (total) | air (INCA) | Public | Health | HI |
| Cd | air (INCA) | Public | Health (carc.) | HI |
| Hg, Pb, HCl | air (INCA) | Public | Health | MI, no DRF |
| As, Be, Cr | air (INCA) | Public | Health | LI |
| heavy metals, dioxins | soil (INCS)) | Public | Health | HI, very complex |
| Sulfates, Nitrates | air (INCA) | Public | Health | HI |
| Ozone (via $NO_x$) | air (INCA) | Public | Health | MI |
| Ozone (via $NO_x$) | air (INCA) | Agriculture | Crop damage | MI |
| CO | air (INCA) | Workers | Health | LI |
| $CO_2$ | air (INCG) | World | Global warming | HI |
| $PM_{10}$, $SO_x$, $NO_x$ | air (INCA) | Agriculture | Crop damage | MI |
| $SO_x$ $NO_x$, HCl, HF | air (INCA) | Buildings | Materials damage | MI |
| $SO_x$, $NO_x$, HCl, HF | air (INCA) | Forests, lakes | Acidification | LI |
| HCl, HF | air (INCA) | Workers | Health | LI |
| pCDD, pCDF | air (INCA) | Public | Health (cancer) | HI |
| PAH, PCB, HCHO | air (INCA) | Public | Health | MI, no data |
| Construction | other (INCB) | Workers | Accidents | MI |
| Operation | other (INCB) | Workers | Accidents | LI |
| MSW transport | congestion, air (INCA, INCO) | Public, Workers | Odours, traffic | MI |
| Visual impact | (INCV) | Public | Amenity | HI |
| Visual impact | (INCV) | Residents | Estate value losses | HI |
| Visual impact | (INCV) | Public | Visibility | LI |

**Table 1.3b.** Summary of expected burdens and impact pathways

| Burden | Pollution vehicle | Target | Impact | Expected relevance and difficulties |
|---|---|---|---|---|
| **Disposal of solid incineration residuals (ASH)** | | | | |
| Leachate leakages and treatment cycle | soil (ASHS) | Public | Health | HI/LP |
| Leachate leakages and treatment cycle | soil (ASHS) | Wildlife | Food chain | HI/LP |
| Windblown bottom ashes | air (ASHA) | Public | Health | LI, no data |

**Table 1.3c.** Summary of expected burdens and impact pathways

| Burden | Pollution vehicle | Target | Impact | Expected relevance and difficulties |
|---|---|---|---|---|
| **Landfill disposal of MSW (LFL), including biogas collected (COL) and dispersed (DIS)** | | | | |
| $CO_2$ | air (DISG,COLG) | World | Global warming | HI |
| $CH_4$ | air (DISG) | World | Global warming | HI |
| Water steam | air (DISG) | World | Global warming | LI |
| Cl - compounds | air (DISG) | World | Ozone layer Depletion | LI |
| $H_2S$, S-organics | air (DISA) | Forests, lakes | Acidification | LI |
| $H_2S$, S-organics | air (DISA) | Public | Odours | HI |
| Ozone (via $NO_x$) | air (INCA) | Public | Health | MI |
| Ozone (via $NO_x$) | air (INCA) | Agriculture | Crop damage | MI |
| PAH | air (DISA) | Public | Health (carc.) | MI |
| $SO_2$, $NO_x$ | air (COLA) | Public | Health | MI |
| Leachate leakages and treatment cycle | soil (LFLS) | Public | Health | HI/LP |
| Leachate leakages and treatment cycle | soil (LFLS) | Wildlife | Food chain | HI/LP |
| Fires | air (LFLA) | Public | Health | HI/LP |
| Visual impact | (LFLV) | Public | Amenity | HI |
| Visual impact | (LFLV) | Residents | Estate value losses | HI |
| MSW transport | Congestion, air (LFLA, LFLO) | Public | Odours, traffic | MI |
| Pathogenic dissemi-nation by animals | other (LFLO) | Public | Health | LI/MI, no data |
| Construction | other (LFLB) | Workers | Accidents | MI |
| Operation | other (LFLB) | Workers | Accidents | LI |

Notes:
HI/MI/LI = high/medium/low expected impact; LP = low probability; DRF = Dose-Response Function; carc. = carcinogenic; -A = air pollution; -G = global impacts; -S = soil and groundwater pollution; -V = visual impact ; -B = impact on plant builders; -O = other impacts

Specific pollutants responsible for health effects can be divided into *carcinogenic* and *non-carcinogenic*.

Cd and pCDD-pCDF belong to the former group. They are supposed to cause cancer, respectively, to lung and liver (Berrino et al., 1995). Although debate is open on the shape of their dose response function, most researchers now tend to consider the probability of incurring a cancer as linearly increasing with the dose inhaled.

Hg, Pb and HCl, on the other hand, belong to the latter group. Mercury can damage human liver, kidney, and also brain tissues. Lead brings about neurological effects (saturnisms), and is known to permanently lessen brain capacities. HCl does not accumulate in the human body; if present in low concentration, like those of the incinerator case, it may just lead to possible minor damages to the workers (eyes and throat irritation, as we said before).

Usually, human health risk evaluations referred to these *non-carcinogenic* pollutants do not make use of any dose-response function. A frequent kind of evaluation is based in this cases on an *index risk*, which represents the ratio between the measured ground concentration and a given maximum acceptable concentration level. Our analysis will only provide a "dichotomic" information, reporting whether the ground level concentrations exceed the threshold value identified.

Impacts of macro pollutants on global warming (GW) and on agriculture (crops) are closely similar to those arising from fossil fuel cycles so that their evaluation will follow the same approach as in other ExternE studies and the methodology will not be described in details in this report (for a deeper description the reader should refer to EC - DG XII, 1995b).

A site-specific analysis of materials corrosion in buildings and monuments would be desirable due to the presence of acid gas emissions, but it was impossible to find or produce reliable national data on targets. These impacts are therefore estimated by means of the database contained in the Ecosense software. The reader should however be aware, when reading our results, that this database may not adequately represent the material composition of the Italian buildings.

Visual impact and other holistic valuation has been developed by hedonic techniques; the model used to evaluate local odours dispersion is described in section 3.6.

### 1.5.3 Priority impacts of landfill sites

In the case of landfill sites, most impacts differ to those of incineration as well as from fossil fuel cycles. Exceptions are represented by emissions of exhaust gases from biogas combustion and GW impacts of carbon dioxide and methane that will be estimated with methods close to those implemented for other fuel cycles. Note however that landfill gas impacts on health and global warming require special care in the evaluation stage if accounted as avoided impacts of incineration, as they display a delayed time path with respect to electricity generation (see section 2.2).

Specific impacts of aromatics on health are considered by looking at benzene. While this is by no means the only dangerous pollutant arising from landfills, the actual composition of trace substances found in landfill gas is highly variable and depends on waste composition and landfill chemistry in ways that are not known with certainty. It was decided to consider only one of the worst and most common pollutants as a "tracer" of dangerous organic compounds.

Likewise, impacts of odours could be attributed to several (mainly sulphur-organic) compounds. Methanthiol is usually chosen as an appropriate tracer for this class of impacts.

$NO_x$ and $SO_2$ emitted by biogas combustion must be treated just as those arising from incineration.

As in that case, only direct (inhalation) pathways are considered in the evaluation of health effects. The same reservations outlined in the previous section do apply.

Let us turn to impacts of the landfill site arising through soil and groundwater contamination. Their likely health impact through both direct and indirect pathways might be as relevant as that of atmospheric pollution in case the landfill is not provided with the necessary equipment to prevent escape; however, as we assume that the landfill operators follow the law, we may well consider this impact pathway as not particularly relevant for our analysis.

There is furthermore another reason for not considering such pathways within this study. As stressed in section 1.2.1, we are interested in the assessment of externalities as defined by equation (3) and (4) of the same paragraph, that is those of incineration and disposal of its residual *less* those of avoided landfilling of MSW. By considering Table 1.3 (excluding minor impacts), these can be split into several classes:

$$EXT = INCA + INCG + INCV + INCB + ASHS - (LFLA + LFLG + \\ + LFLV + LFLB + LFLS + LFLO) \quad (4)$$

Now, even though ASHS and LFLS were quite serious, the impact of their difference would probably be much smaller. A large part of the impact derives from contamination of soil and groundwater by heavy metals, but that would happen both in MSW landfill or in a plant for disposal of incineration residuals. In fact, as shown in Brunner-Monch (1986), most heavy metals are dispatched to the landfill site anyway (as slag, trapped in the electrostatic precipitator, or dispersed in the untreated MSW), with the exception of mercury and to some extent cadmium.

On the other hand, slightly smaller amounts of heavy metals in ASHS with respect to LFLS must be balanced against highest volatility of the ashes and highest concentration of heavy metals in landfill leachate. Evaluation of this difference is hardly worth pursuing if we consider the large difficulties involved therein. Thus, we assume

$ASHS - LFLS \cong$ negligible.

No other class of externalities can however be ignored. Effects on global and local atmosphere, visual and odour impacts and accidents of the construction stage largely differ between MSW incinerators and landfills. Unfortunately, as stated above, some other effects like hygienic consequences of landfills due to pathogenic dissemination by birds, mice and other animals (LFLO) could not be reasonably estimated.

### 1.5.4 Priority impacts of MSW transport

As regards externalities of transport, according to what has been explained in section 1.2.1, they will be evaluated in terms of the *net* impact arising from the fact that landfills are usually located further than incinerators from the town producing wastes.

The impact of transport is mainly due to atmospheric pollution and road accidents. The emissions of a diesel lorry transporting MSW contain relevant quantities of all major macropollutants: particulate, sulphur oxides and nitrogen oxides, as well as carbon monoxide. It is important to notice that the greatest part of particulate matter emitted by diesel vehicles is of a very small dimension, i.e. it belongs to the $PM_{2.5}$ group. This is supposed to be more dangerous (i.e., exposure-response functions related to it take higher values) than the coarser PM ($PM_{10}$).

In addition to these macropollutants, transport emissions contain also considerable quantities of Volatile Organic Compounds. Some carcinogenic substances belong to this group, and would deserve a careful consideration. However, it is still quite difficult to know precisely what are the substances contained in VOC transport emissions, and in what proportion they are (see e.g. Bailey et al., 1990).

The impact pathways concerning the atmospheric pollution coming from transport emissions show some distinguishing features as compared to those considered for the same substances in the case of power generation plants. Firstly, transport emissions are produced by mobile sources. This involves the use of different dispersion models to evaluate the corresponding change in ambient concentrations. Secondly, the emission rates are much more variable than in the power generation case, since they depend not only on the particular "plant" (i.e. vehicle) considered, but also on the way this particular vehicle is run (what is called *driving cycle*). Care has thus to be put in specifying the driving cycle considered in the analysis. Finally, they are produced from very small and low "stacks", only a few centimetres high above the ground. For this reason, pollutants are likely to remain within an area much closer to the source - the "regional" damage is expected to be considerably lower, or even negligible, as compared to the "local" damage, as the ExternE-Transport project has shown (EC DG XII, 1997).

### 1.5.5 Locations.

The typical size of waste incinerators and landfills is in the order of some hundreds of tonnes of MSW disposed per day. While some large cities run larger plants (above 1000 t/day) it is expected that future plants will be smaller in order to reduce and diffuse environmental impacts, curb transport costs and possibly reduce social opposition. Several European countries have recently passed legislation requiring that MSW treatment and disposal be mostly "local", i.e. within the boundaries of such constituencies like the UK or Irish *County*, the French *Département* or the Italian *Provincia.* Since:

- the average population of these administrative units is usually in the order of 600,000;
- MSW per capita production rates in Europe are in the order of 1 kg/day

- it is increasingly unlikely to find a single technology (and therefore a single plant) within each of them due to lower costs of landfilling and current policies favouring waste avoidance and recycling; it may then be expected that typical plants to deal with current MSW production in Europe should be around 100-500 t/day. On the other hand, too small plants should be avoided due to increasing capital costs; incineration in particular is characterised by remarkable economies of scale.

With such an average plant size, the technology examined in this study should not be regarded as particularly widespread, although it may well represent the reality of large urban areas, as that of Milan.

The study on the externalities of energy recovery from MSW (unlike those on large power stations) would require the implementation of the same exercise for a large number of sites and plants in order to give more precise information for the purpose of accounting for their external costs; nevertheless, even the analysis carried out on a single site can be extremely useful to the policy maker in charge of selecting technological options to deal with wastes and save energy.

Moreover, it has to be emphasised that the two sites chosen for the incinerator and the landfill analysis can be considered highly representative of situations easily found in the Po valley in Northern Italy, where a population of about 20 million live and where conditions of meteorology, agriculture, urban and social framework are relatively similar. The exercise could therefore set the methodology and provide the framework of environmental data useful for future analysis of actually planned facilities. Caution should be used in any attempt of transferring the external costs evaluation to other sites, especially as regards the estimate of amenity losses, which may be more site-specific than other results.

As regards the incinerator site, the projected - but still not realised - plant of via Silla is located in the western outskirts of Milan, where the population density is high (the average for the whole town is around 8000 inhabitants per square km). As often happens the incineration plant, which needs a smaller space to be built compared to the landfill, can be located closer than the landfill to the place where wastes are originated. This allows savings in the transport costs (both "internal" and external), but may also cause on the other hand an increase in the external costs from power generation, due to the fact that the stack is placed in the middle of a densely populated area.

The reference case chosen for the landfill is the now exhausted and closed site at Cerro Maggiore, about 25 km NW of Milan. For several years before its closure this site has been used for disposal of most of the wastes coming from Milan. Although this location is usually defined as rural, the population density in this area is quite relevant, reaching an average of about 1800 inhabitants per square km. This relative similarity of meteorological characteristics and of lifestyle patterns across the Po valley should ensure a fair degree of transferability of our estimated damage functions to most sites in medium to high population density areas of Northern Italy.

# 2. ASSESSMENT OF PHYSICAL BURDENS

## 2.1 Evaluation of emissions from a MSW incinerator

As mentioned above, macropollutant emissions from an incinerator stack consist mainly of dusts, nitrogen and sulphur oxides, total organic carbon, carbon oxides, and hydrohalogen acids; problems of micropollutants arise on the other hand from the fact that the emissions contain classes of compounds such as heavy metals (chiefly cadmium, lead and mercury), organochlorinates and polycyclical aromatic hydrocarbons.

After MSW combustion, toxic micropollutants contained in wastes can remain as bottom ashes, dust trapped by air pollution controls, wastewater, and fly ashes or flue gas pollutants not trapped by air pollution controls.

The first two are not likely to bring about considerable risk for human health, provided that they are correctly disposed of in suitable landfills; their treatment costs are therefore internalised. The same may apply to wastewaters if they are properly treated and slags are properly conferred to sanitary landfills. Such materials are likely to bring about environmental problems with external costs only in case of accidents occurring to the landfill walls, or possibly during their transportation to landfills due to dust which could be windblown. However, we were not able to find any study trying to estimate neither the probability nor the quantity of dust windblowing during transportation. Anyway, the quantities involved are likely to be of very small concern, and can reasonably be neglected.

**Table 2.1** Incineration residuals (source: Giugliano et. al, 1991)

| | | |
|---|---|---|
| Bottom ashes | 280-350 | kg/t |
| Fly ashes | 25-30 | kg/t |
| Slag | 1-3 | $m^3/t$ |
| Wastewater | 1-5 | $m^3/t$ |

It always turns out to be very difficult to predict with a good degree of approximation the quality and the quantity of specific pollutants emitted from an incinerator stack. In fact, they heavily depend on the chemical composition of wastes which are burned and on technological features of the incinerator. Further, it is known that both waste composition and incineration technology show a huge variability either in space (depending on the particular country and region it is considered) or across time (in fact, waste "quality" changes with consumption

habits). Due to this variability, there results a very large number of toxic pollutants which can be potentially emitted by an incinerator in some variable quantity.

Since the incineration plant considered here, has not been built yet, the emission factors taken in the analysis reproduce the emission performances guaranteed by the constructor who won the bid for the plant realisation. Where emission factors for single pollutants were not available (some heavy metals emissions are usually grouped together, so that emission factors a single element cannot be known) we considered representative emission factors taken from the literature.

Table 2.2 presents an estimation of their flows as contained in a sample of incineration residuals.

All emission rates listed in Table 2.4 fall below both the EU (Directive 369/89) and the Italian legal standards.

**Table 2.2** Metal flows in the incineration residuals (source: Giugliano et. al, 1991)

| | Bottom ashes | Fly ashes | Wastewater |
|---|---|---|---|
| | g/kg | g/kg | g/kg |
| Cd | 0,0079-0.02 | 0.30-0.83 | <0.02-0.05 |
| Cr | 0.4-0.7 | 0.82-1.06 | <0.05-0.1 |
| Cu | 1.3-2.0 | 1.9-3.23 | <0.05-0.1 |
| Hg | <0.003 | <0.003 | <0.05 |
| Mn | 0,61-1.0 | 0.63-1.0 | <0.03-0.08 |
| Ni | 0.19-0.54 | 0.26-0.35 | 0.07-0.2 |
| Pb | 0,99-2.1 | 7.5-18.1 | 0.3-0.7 |
| Zn | 1.87-4.6 | 20.1-27.0 | 0.12-0.37 |

**Table 2.3** Technical features of the Milano – Silla incinerator

| | |
|---|---|
| Capacity | 900 t/day |
| Working days | 310 days/year |
| Net calorific value | 15000 kJ/kg |
| Energy efficiency | 0.21 |
| Electricity output | 247.7 GWh/year |
| Flue gas temperature | 130° C |
| Flue gas production | 250,000 $Nm^3/h$ |
| Stack height | 100 m. |
| Stack diameter | 3 m. |
| Waste composition (in weight) | |
| Paper and cardboard | 31% |
| Organic matter | 29% |
| Plastic | 17% |
| glass and inert matter | 2% |
| wood | 1.4% |
| metals | 0.4% |
| clothes | 5.2% |
| napkins | 3.5% |
| other | 10.5% |

No legal limits exist of course for $CO_2$ (apart from international voluntary agreements to reduce greenhouse gases emissions, which do not concern *individual* plants). Emissions of carbon from incineration of MSW is estimated at 0.3 tonnes per tonne of waste by Wallis (1994), while UKDOE (1993) suggests 0.17-0.22 for a mix of solid wastes of which MSW are probably the largest component. This coefficient can hardly be evaluated with greater precision as it depends mainly on the carbon content of wastes. The Wallis estimate was accepted as more appropriate for MSW; it amounts to 1.1 tonnes ($CO_2$)/tonne (MSW) or (under the above technical assumptions) 1.2 kg ($CO_2$)/kWh, which is 20-30% higher rate than those typically generated by coal-fired power stations.

**Table 2.4** Emission rates of the incinerator

| Pollutant | Unit | Emission rate |
|---|---|---|
| PM | $mg/Nm^3$ | 3 |
| $NO_x$ | $mg/ Nm^3$ | 200 |
| $SO_2$ | $mg/ Nm^3$ | 50 |
| CO | $mg/ Nm^3$ | 50 |
| HCl | $mg/ Nm^3$ | 10 |
| HF | $mg/ Nm^3$ | 1 |
| PAH | $mg/ Nm^3$ | 50 |
| Cd | $\mu g/ Nm^3$ | 10 |
| Pb | $\mu g/ Nm^3$ | 5 |
| Hg | $\mu g/ Nm^3$ | 20 |
| TCDDe | $pg/ Nm^3$ | 50 |

Emission data and measurement issues deserve some further comments.

First of all, as we said before, it is not easy to find accurate and representative data on average emissions of those substances which are emitted just in traces and for which there is no legal standard requiring frequent and accurate measurements.

Moreover, it is often very difficult to detect the presence of some particular trace pollutants in flue gas emitted. This is the case e.g. of dioxins and furans, which are emitted in very low quantities.

In addition to that, measurement are often lacking because of the fact that many toxic elements are measured in their *total* emissions, regardless of the specific chemical form in which they were found at the stake. As example, metals can come out as oxides, or combined with some non-metals to form a particular salt. However, pollutant toxicity for human health is often dependent on the chemical form in which the pollutant leaves the stack and reaches human receptors.

A basic inconsistency may arise between emission data, where toxic pollutants are in some way grouped, and dose response functions, which are frequently specified for particular

compounds belonging to the same group. Therefore, a greater precision in emission measurement should be required for the analysis to be more accurate. However, greater precision in measurement and data collection can be obtained only with a considerable increase in costs and in time needed for measurement to be completed. On the other hand, the opposite case may also occur as scientists may ignore the toxicity of each specific chemical compound even though information exists for the whole group of compounds.

A cautious, acceptable solution to this problem could be that of considering all pollutant compounds belonging to the same group as having *the same toxicity*, and to derive a global average estimate of environmental risk (or an upper/lower bound estimate, depending on which toxicity level is chosen).

As for the particular case of dioxins and furans, which owing to their high toxicity have been deeply analysed, estimates are known concerning the relative toxicity of different compounds belonging to the same group (pCDD-pCDF). There are 75 different isomers belonging to the pCDD group, and 135 belonging to that of pCDFs. All of them show similar chemical properties and are thought to be toxic, but the toxicity factor is not the same for every compound of each group. The International Toxicity Equivalency Factor (I-TEF) scale (see Table 2.5 below), which is now one of the most used, was applied to give emission data directly in terms of 2,3,7,8-Tetra-Chlorinated-Dibenzo-Dioxins equivalent (TCDDe), that is in terms of the most dangerous dioxin congener.

Some difficulties may also arise for macropollutants, in particular as regards $PM_{10}$. In fact, the most dangerous particles are those with smaller dimension, which are more likely to get over human body's defences and to damage the respiratory system. Epidemiological studies (and particularly those to which the ExternE set of dose-response functions is referred) often consider those whose diameter is less then 10 micron, indicated as $PM_{10}$. However, there is a lack of precise data concerning the exact proportion of $PM_{10}$ for incinerator emissions. The USA standard conversion factor assumes that $PM_{10}$= 0.55 TSP. However, this coefficient is quite likely to underestimate average $PM_{10}$ incinerator emissions. Data reported in Radian Corporation (1989) on particle size distribution in particulate matter, based on tests run for some US incinerators, show a $PM_{10}$ proportion of 65.7 to 86.2%. This data can be highly specific for the particular air pollution control technology, and for the particular quality of wastes burned in the tested plants. Due to the high variability of the incinerators TSP dimension and to the lack of specific data for the plant evaluated, we chose to use in our analysis the conservative assumption that all TSP emitted from the incinerator belong to the finer particulate ($PM_{10}$).

As regards atmospheric pollutants dispersion on both the local and regional range, the study on the incinerator impact totally relies upon the use of models contained in the Ecosense 2.0 package (EC DG XII, 1995b; Krewitt, 1997).

**Table 2.5** International Toxicity Equivalency Factors (I-TEF) Method of Risk Assessment for Complex Mixtures of Dioxins and Related Compounds. (Source: NATO-CCMS, 1988)

| *Congener of concern* | *I-TEF* |
|---|---|
| 2,3,7,8, TCDD | 1 |
| 1,2,3,7,8,PeCDD | 0.5 |
| 1,2,3,4,7,8 HxCDD<br>1,2,3,7,8,9 HxCDD<br>1,2,3,6,7,8 HxCDD | 0.1 |
| 1,2,3,4,6,7,8 HpCDD | 0.01 |
| OCDD | 0.001 |
| 2,3,7,8 TCDF | 0.1 |
| 2,3,4,7,8 PeCDF<br>1,2,3,7,8 PeCDF | 0.5<br>0.05 |
| 1,2,3,4,7,8 HxCDF<br>1,2,3,7,8,9 HxCDF<br>1,2,3,6,7,8 HxCDF<br>2,3,4,6,7,8 HxCDF | 0.1 |
| 1,2,3,4,6,7,8 HpCDF<br>1,2,3,4,7,8,9 HpCDF | 0.01 |
| OCDF | 0.001 |

## 2.2 Evaluation of emissions from a MSW landfill

The scant availability of data on the extent and chemical nature of emissions from dumps allows only indirect evaluations based on presumable losses into the atmosphere of gas produced by the degradation of waste. Such losses moreover evolve over time in relation to the degree of filling of the dump, composition and transformation of waste, landfill imperviousness, moisture and temperature. Moreover, landfill gas emission tend to increase with better soil isolation.

Biogas generation reaches its maximum within a few years after the refuse has been deposited and compacted in the dump, in the order of 15-20 cubic meters per tonne of MSW/year. This emission rate has then a tendency to decline down to 3-7 $m^3$/tonne/year for a plant aged 10-20 years (Cernuschi, Giugliano, 1989; Stegmann, 1988; Willumsen and Bach, 1991). Total

emission arising over time from one tonne of MSW are estimated by UKDOE (best) at 133 cubic meters.

This study evaluates an emission hypothesis based on differentiated productions of gas according to the age of the waste, and on the systems of gas collection, active only in the completed and covered parts of the sites. For this purpose the landfill site is envisaged capable of dealing with 2000 tonnes/day for 10 years (although the real site evaluated has been working for a slightly shorter period).

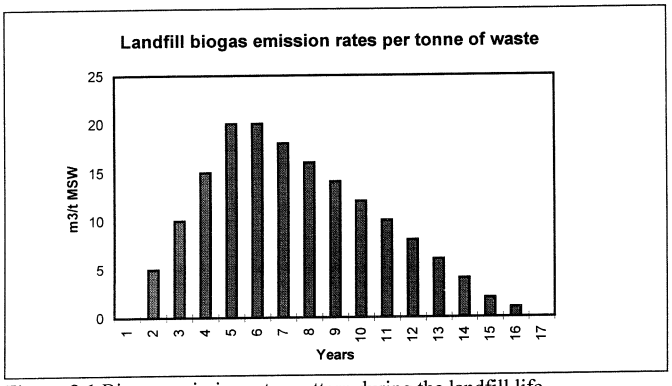



**Figure 2.1** Biogas emission rates pattern during the landfill life

Landfill gas production from each unit of disposed refuse begins about one year after the start of site cultivation and increases steadily up to a maximum of 20 $m^3$/tonne in the fifth year; after that time it declines linearly until it vanishes in the fifteenth year (Figure 2.1). Therefore, biogas production for the whole site is assumed to last 24 years and totals 160 $m^3$/tonne of MSW.

The relatively high emission rates in the early years are also meant to account for lower collection rates obtaining in the filling stage. The biogas collection rate is assumed instead to be constant throughout the life of the site and set to 60%.

Gas emission are assumed to be uniformly distributed over the whole area of the landfill site.

Landfill gas is made mostly of carbon dioxide and methane. Theoretically, their production should be equivalent (50% each), but in fact it varies with the emission rate. On average, 55% of $CH_4$ and 45% of $CO_2$ are reported: these rates will be assumed to be constant throughout the life of the plant, without relevant loss of information. Nitrogen may be contained in the gas due to infiltration into the landfill.

In evaluating the impact on the area caused by odours and toxic substances transported with the dispersed gas, the acquisition of a series of analyses of trace substances carried out on the

gas in urban landfill sites of different ages (Young-Parker 1983; Walsh et al 1988) were taken into account. Among the more than 100 components present in trace quantities *benzene* was identified to represent the emission situation with regards to toxic substances. The criterion of choice of trace substances was based on the maximum value of the ratio between the concentration with which the trace substance is contained in the gas, and its TLV (threshold limit value). The TLV is as proposed by the American Conference of Governmental Industrial Hygienists (ACGIH - as in Radian Corporation, 1989). With regard to toxic substances the dominant tracer is *benzene*. Each cubic meter of landfill gas is assumed to contain 17 mg of this tracer. The impact of toxics is traced by benzene emission.

The dispersion pattern of benzene has then been analysed applying the usual ISCLT (Industrial Source Complex Long Term) model and WTM (Windrose Trajectory Model) used in the Ecosense 2.0 package (Krewitt W. 1997).

Contrary to that, the dispersion and impact of malodorous substances on the local scale was considered by using a different, more detailed atmospheric dispersion study, which is specifically addressed in section 3.6.

About 60% of the landfill gas generated is collected and burned in four Otto-cycle engines to raise electricity. The remaining 40% is directly dispersed into the atmosphere.

Flue gases produced after combustion are conveyed into a 50 m high stack with a diameter of 0.6 m. About 33,330,000 $Nm^3$/year are emitted through the stack. The average flue gases temperature is around 770 K.

**Table 2.6** Estimated emission rates of the landfill site (Cerro Maggiore)

| **Pollutant** | **Flue gas** | **Dispersed biogas** |
|---|---|---|
| | $mg/Nm^3$ | $mg/Nm^3$ |
| $PM_{10}$ | ng | ng |
| $SO_x$ | 55.3 | ng |
| $NO_x$ | 247.9 | ng |
| Benzene | ng | 17 |

Among macropollutants contained in flue gases, $NO_x$ is the most important emission (in terms of quantity); $PM_{10}$ emission rates are rather negligible. For other toxic substances contained in the biogas - in particular organic micropollutants - they are assumed to be destroyed by the combustion processes due to the very high temperatures reached inside the engine. $CO_2$ and methane emissions are further specified in section 3.8.

As for the incineration plant, the impact of landfill emissions was evaluated according to the Ecosense 2.0 models.

## 2.3 Evaluation of emissions from MSW transport

Once deposited into a provisional store, where they are conveyed by specific waste collection vehicles, municipal solid wastes are transported to the treatment plant by a diesel, heavy duty vehicle, with 15 tonnes capacity. Given this capacity and the amount of wastes yearly disposed of in the landfill, a total number of just under 42,000 trips are necessary, each of 25 km length (plus an equal length for the way back). This amounts to an overall distance of about 2,080,000 km covered per year.

The emission rates for the vehicle considered were taken from the Corinair (1991) report. Pollutants considered were the "traditional" macropollutants ($PM_{10}$, $SO_x$, $NO_x$, CO), as well as $CO_2$ and VOC (among which some carcinogenic micropollutants were specifically addressed).

Since this is a *mobile* source of atmospheric emissions, as mentioned above, the dispersion pattern could not be properly evaluated by using Ecosense 2.0, particularly as regards the *local* results. According to the methodology defined in the ExternE-Transport project (EC DG XII, 1997), a different package was identified in ROADPOL and applied to this evaluation. A detailed description of this model is provided in the ExternE -Transport report (EC DG XII, 1997), and will not be repeated in these pages.

**Table 2.7** Heavy duty diesel vehicle emissions according to Corinair (1991)

| Pollutant | Emission factor (g/km) |
|---|---|
| $PM_{10}$ | 0.82 |
| $SO_x$ | 0.15 |
| $NO_x$ | 7.4 |
| CO | 7.3 |
| VOC | 0.76 |
| $CO_2$ | 590 |
| Fuel Consumption (g/km) | 189 |

Meteorological input data for the ROADPOL long term dispersion model consist in the joint frequency distribution of wind speed, wind direction and atmospheric stability class. All data utilised in the evaluation were taken from Milan Linate meteorological station. Mixing heights range between 194 and 630 m, depending on atmospheric stability class.

A Cartesian grid of 40 x 40 km was defined to locate line emission source and receptors. Taking a step of 2 km for both axes, 441 receptors have been located at the intersection of all co-ordinates. The ROADPOL-Long Term model has then been applied for evaluating ground level concentration of the pollutants considered for each receptor. The analysis also considered a larger area, covering a 100 x 100 km square with 5 km step for receptors. This latter estimation, however, did not add any relevant information to the former one, as 99.9% of the impact lies within the 40 x 40 km square.

Correction for low wind speed were not taken into account, even if these speeds are very frequent at the site analysed. Due to the large extension of the area considered dispersion parameters have been calculated by Pasquill-Gifford's formulations. This combination of ROADPOL-Long Term model parameters resulted in a rather conservative estimation of ground level concentrations.

Ground concentration data were finally combined with population, in order to obtain *weighted pollutant average concentration* defined in the usual formula (see e.g. ORNL 1991):

where $P_i$ and $C_i$ are population and pollutant ground concentration data of the i.th gridcell.

$$C_{pop}(.) = \sum_i P_i \cdot C_i(.) \Big/ P_{tot}$$

The average concentration level defined above was then linked to an Exposure-Response Coefficient (ERC) and to the total population number (Ptot) in order to find total damage (D) in physical units, as in the formula below:

$D = C_{pop} * ERC * P_{tot}$.

It should be recalled that this use of an *average* ground concentration level is allowed without any information loss only in case of linear, "no-threshold" ER Coefficients (as those used in this and most other ExternE studies). Otherwise, in case the Exposure-Response relation is given by a non-linear function $ERF_i = F(C_i)$ , total damage D ought to be calculated as a total sum of single gridcell damages.

# 3. ASSESSMENT OF IMPACTS AND DAMAGES

## 3.1 Identification of targets

The "targets" of the various impact pathways considered in this analysis are essentially population (the damage to which has shown to account for the bulk of the external costs in fossil fuel power generation plants), crops, forests and materials (buildings). They were considered both at the regional and local scale. Data about the distribution of these targets around the plant are contained in the Ecosense 2.0 database. This database assumes an homogeneous distribution of the targets within each gridcell. The grid resolution is 100 x 100 km as concerns the analysis at regional scale (the whole Europe). A finer resolution of 10 x 10 km for each gridcell is considered at the local scale, which is defined as the 100 x 100 km square centred on the plant.

An even finer resolution (2 x 2 km), as mentioned before, was necessary to evaluate local MSW transport damage. This is because of the low height of the atmospheric emission source, which renders more limited the spatial diffusion of pollutants, and makes it worth to carry out a more detailed analysis on a shorter range. The data on the distribution of various targets subject to pollution from MSW transport were taken from Istat 1981, 1990, 1991, 1992. It is worth mentioning that the whole local area where MSW transport takes place is characterised by a relatively high population density, so high that is sometimes considered as belonging to an "extended" metropolitan area of Milan. For this reason, the quantity of crops - mainly rice and maize - and forests present on it is relatively small. Conversely, a relevant presence of buildings should be expected in the area. However, no local specific data on the number and material composition of buildings were available for the implementation of the analysis on the local range. Therefore, Ecosense estimates on the number of buildings and on their material composition were used. The reader should be warned that these Ecosense estimates are based on studies made exclusively on Northern European samples, and hence they are not particularly representative of the Southern European realities. A high degree of uncertainty is thus likely to affect this part of the estimation.

## 3.2 Methodological Issues

Physical and monetary evaluations of most of the impact pathways hereby considered strictly follow the ExternE methodology, by applying wherever possible the same (Ecosense 2.0) dispersion models, the same ERF (ERF related to public health are reported in details in Appendix 4) and the same monetary values (Table 3.). Therefore, these methodological aspects will not be further described here. The reader can refer to EC DG XII (1995b) and to EC DG XII (1997) reports for more details.

Two issues, on the other hand have not been particularly considered by the other ExternE reports, and will be better analysed in this work, being a more peculiar aspect of the incinerator "fuel cycle". The first issue, described in sections 3.3.1 and 3.3.2, is the set of impact pathways given by atmospheric *micropollutants*. The second issue, treated in the whole section he evaluation of disamenities of waste disposal site, is given by the evaluation of the disamenities of waste disposal sites.

## 3.3 Impact of Micropollutants

### 3.3.1 Direct pathways

A strongly negative characteristic that most micropollutants show is that of being *persistent*, i.e., they cannot be reduced to a harmless state by natural agents (heavy metals) or they can be reduced but only after several decades (dioxins and furans). Therefore, not only can they accumulate in the environment, but they are also transported and accumulated (by atmospheric agents, or through the food chain) in some unpredictable ways, giving rise to potentially great dangers at a local level.

Due to this characteristic, a fundamental distinction between *direct* and *indirect* impact pathways has been introduced. *Direct* impact pathways are those followed by air pollutants damaging human health "directly" through air inhalation. *Indirect* impact pathways, on the other hand, are those occurring after pollutant deposition on soil. In this case, man can be exposed to these pollutants by ingestion (of contaminated soil risen up by the wind - mainly for children -, and of contaminated food or water) and/or dermal contact, as will be explained below.

Another basic distinction which has to be introduced, as regards the specific health effects that the pollutants listed above (both organic and inorganic) can cause, is that between *carcinogenic* and *non-carcinogenic* substances.

Cadmium and dioxins/furans belong to the former group. They are believed to cause cancer, respectively, to lung and liver (F. Berrino et al., 1995; L. Tomatis, 1991). In both cases, unfortunately, survival chances are very low. Survival percentage for lung cancer is around 8% after 5 years (*ibidem*). It is not possible to find statistical evidence of survival specifically for liver cancer, since there are some difficulties arising in its diagnosis, but it is also likely to be very low.

Although there is still open debate on the shape of carcinogenic dose response functions, most researchers tend actually to consider the probability of incurring a cancer as linearly increasing with the dose inhaled, *without any threshold effect*.

On the other hand, mercury and lead belong to the latter group. Mercury can damage human liver, kidney, and may also accumulate in brain tissues injuring them. Lead brings about

neurological effects (saturnism), and is known to permanently reduce brain capacities, particularly in children. Unlike carcinogenic pollutants, curves linking human exposure to these pollutants to the probability of incurring health impacts are thought to show threshold levels. In other words, a threshold level below which no negative effects occur is supposed to exist.

Usually, for this reason, human health risk evaluations for these *non-carcinogenic* pollutants do not use dose-response functions. The most frequent kind of health risk evaluation is based in these cases on a *risk index.* This is defined as the ratio between the measured ground concentration and a given maximum acceptable (harmless) concentration level.

In our case, every gridcell belonging to the area considered shows concentration levels (estimated by Ecosense 2.0) smaller than the Pb and Hg impact thresholds defined, therefore such impacts are given a zero value for this case.

The same distinction between *carcinogenic* and *non-carcinogenic* pollutants holds independently of the specific pathway considered.

Direct impact pathways have been studied and quantified not only for macropollutants generated by traditional power plants, but also for some micro pollutants emitted by waste incineration processes. For carcinogenic pollutants (Cd and dioxins) we took Exposure-Response Functions based on the concept of Unit Risk Factors (URF) described e.g. in Radian Corporation (1989).

URF is defined as an estimate of *the probability that an individual will develop a cancer when exposed to a pollutant at an ambient concentration of one microgram per cubic meter for a lifetime* (which is assumed to be 70 years) (See also EPA, 1990).

Unit Risk Factors are quite different from the exposure-response coefficients considered for macropollutants. The latter are based, as we said before, on elaborations of epidemiological studies. On the other hand, URFs usually hinge either upon specific experiments run on animals (in particular on mammals, since they are more likely to show responses similar to those of man), which are exposed to high micropollutant doses for a given time period, or upon some observed effects on human health (e.g. in case of accidents). In this latter case, dose-response functions are deducted from *high* exposure levels, and have then to be extrapolated for lower exposures. A linear, no-threshold shape of the functions is generally assumed.

These URFs are less prone to multicollinearity problems than ERFs listed above and used in this study. On the other hand, they also can lead to bias in human health effect estimation, since there are obvious problems both in transferability of URFs from animals to man, and in the extrapolation of dose-response curves to levels as low as those usually involved when average ground concentration levels are considered.

Since we need to establish external effects on an annual basis, URFs were then simply divided by 70 in order to obtain an annual cancer incidence. This is the usual approach adopted e.g. by Radian Corporation (1989). The LAI (*Länderausschuß für Immissionsschutz*) URF were chosen in particular to derive ERF to be used with Ecosense.

Estimated effects of each pollutant were simply added up in the final step of our evaluation, since nothing is known as regards possible synergistic or antagonistic effects among them. This may well lead to some over or underestimations. There is however no reliable data available to describe the uncertainties that this introduces.

**Table 3.1** Some unit risk factors for carcinogenic micropollutants

| Pollutant | URF | Source |
|---|---|---|
| Cadmium | $1.8 \times 10^{-3}$ per $\mu g/m^3$ | Radian Corporation, 1989 |
| Cadmium | $1.2 \times 10^{-2}$ per $\mu g/ m^3$ | LAI |
| 2,3,7,8 TCDD | 33.0 per $\mu g /m^3$ | Radian Corporation, 1989 |
| 2,3,7,8 TCDD | 1.4 per $\mu g/m^3$ | LAI |
| Benzene | $8.3 \times 10^{-6}$ per $\mu g/m^3$ | NJDEP, 1993 |
| Benzene | $8.0 \times 10^{-6}$ per $\mu g/m^3$ | Pilkington et al, 1997 |

In addition to that, outdoor pollution levels are supposed to be exactly the same as indoor ones. This assumption seems reasonable given the persistence of more of the substances concerned.

In order to account for micropollutant persistency in the environment, it is not enough to consider annual impacts. In principle, the impact of a perfectly persistent pollutant is not finite as it may last forever. However, any attempt of evaluating future effects of persistent micropollutants according to the impact pathways methodology would need to know what is the fate of the substances in the very long run (i.e., whether they remain in the living bodies that they have contaminated, or they return in the environment possibly contaminating other organisms through any direct or indirect pathway). As there are no sufficient information to evaluate long-run impact pathways for these pollutants, our analysis will refer to their yearly damage only. The reader should however be aware that the external cost per mass unit of persistent micropollutants emitted can be higher our estimate due to the reasons mentioned.

### 3.3.2 Indirect pathways

As discussed above, many micropollutants are likely to persist for a long time in the environment, and to be transported in various ways reaching human beings through alternative pathways different from inhalation that we evaluated above.

Pollutants which are more likely to follow the indirect pathways are metals, and some organics for which degradation processes are particularly slow. Among these, dioxins are certainly the most important, but PCB and formaldehyde could also be considered.

Indirect pathways concern every kind of exposure-damage chain (in this case, of human exposure) related to deposited emissions of atmospherical pollutants.

After Atmospheric Toxic and Persistent Pollutants (ATPP) are emitted, they are subject to deposition onto the soil surface. Depositions can be either dry (by sedimentation process) or wet (basically linked to rainfall and humidity). Deposition rate in a given period is largely dependent (as for pollutant atmospheric distribution) on site-specific meteorological data.

Depending on specific soil characteristics of the site considered, deposited emissions can either remain on the surface, or infiltrate into the ground to varying depths. Obviously, these different events give rise to various sets of possible pathways. In a typical urban context, like the close surroundings of the plant of Milano-Silla, ground infiltration of toxic contaminants is quite unlikely, and wind removal of deposited emissions is therefore more likely to happen. On the contrary, in a rural context deposited contaminants will infiltrate with higher probability, and they will then follow very different pathways.

On the other hand, indirect effects of a landfill sites are not considered as most pollutants that could leak from a MSW landfill are not very different from those found in a plant for disposal of incineration residuals. Therefore, changes in total impact that may be attributed to the incineration of combustible fractions of MSW are likely to be negligible as well as very hard to estimate. In other words, however polluting the landfill site may be, impacts through the indirect pathways just mentioned would be nearly the same, as (almost) the same pollution would be generated by the landfill site where bottom ashes and incineration slags are disposed of. Indeed, such residuals would contain most of heavy metals born originally by MSW (and possibly in a more volatile form), with the likely exception of mercury. Lack of specific information on possible differences in the leachability between trace pollutants from MSW and trace pollutants from incineration ashes does not allow for further analysis of differences between landfill sites for MSW and for incineration residuals. Nevertheless, it is reassuring that in both cases such impacts should be very small in the case of modern, isolated landfills.

The indirect pathways from incineration which can damage human health can be split into:
a) ingestion of contaminated food;
b) ingestion of contaminated water;
c) ingestion of contaminated soil;
d) dermal contact.

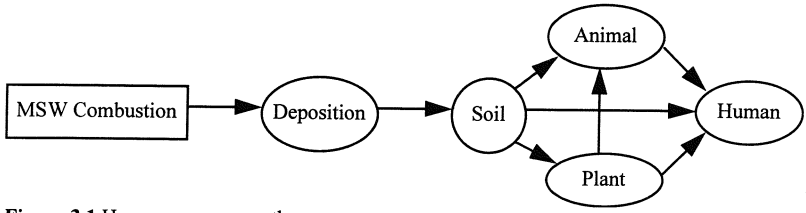



**Figure 3.1** Human exposure pathways

a) Ingestion of contaminated food

Deposited pollutants may contaminate vegetation and animals. Edible vegetables can be contaminated basically in two ways. Firstly, crops cultivated in the polluted area are likely to uptake in their tissues some toxic pollutants deposited on soil (and also those infiltrated in small depth) through root adsorption. The quantities taken up basically depend on the specific kind of pollutant and on the particular vegetable considered. There still remains a large scientific uncertainty regarding plant uptake response to different contaminant kinds and concentrations in soil. Some crops are supposed to linearly increase uptake with pollutant concentration in the soil, up to very high levels without any effect on their appearance. Some others are highly *phytotoxic*, and have a quite low uptake threshold beyond which the plant shows some kind of pathology. Unfortunately, further research is needed to establish with a good order of approximation the most important uptake patterns.

Direct adherence of atmospheric pollutants to plant surface may be another source of vegetable contamination. However, this pathway is likely to involve much smaller quantities than the other one, and it is therefore neglected in most cases.

Animals can also be contaminated either through eating vegetables, drinking water or by dermal contact with contaminated soil. As for the previous case, specific dose-response functions linking pollutant concentrations in soil, water and vegetable tissues with animal tissues uptaking should be estimated and applied to our site-specific case, in order to obtain some rough impact pathway evaluation. However, many of the aspects involved in this pathway have still to be further studied before reliable functions may be assessed.

Finally, and equally difficult, some investigations have to be made on diet composition and other factors of those populations living in the area around the incinerator plant considered. In fact, the health indirect impact analysis requires assessment of an average representative proportion of different kinds of food (vegetables and meat) eaten by exposed individuals, and also an average proportion of food produced in the contaminated area.

A particular dietary pathway, which has been deeply studied, is that of breast milk, in particular for dioxins and furans. In fact, this pathway has shown to be highly efficient in

dioxin transport to breast-fed babies, and should be carefully considered where available data are sufficient to make a quantitative risk estimate.

b) Ingestion of contaminated water

Drinking water can also be contaminated by ATPP emitted by the incinerator. This can occur following two main pathways. Firstly, ATPP may contaminate *surface* water (either by dissolution or by dispersion, depending on the chemical properties of the pollutants considered), mainly in consequence of precipitation events. This constitutes a potential danger for human health, not only if such surface water is utilised as drinking water by some proportion of the residing population (which is not likely to be our case, since almost all drinking water in the area surrounding the two sites chosen comes from groundwater reservoirs), but also in case of some fishermen eat fish caught in contaminated water. Persistent pollutants which are lipophylic, like dioxins, can accumulate in fishes tissues and reach human beings (through the diet pathway) in quite large amounts.

Secondly, and with higher probability in the case hereby analysed, pollutants dissolved in rain or meltwater from precipitation events can infiltrate into the ground and permanently contaminate groundwater reservoirs. However this pathway, which also concerns the landfill risk evaluation, is particularly difficult for analysis and quantification. It heavily depends on soil characteristics, and requires a deep understanding of the probability and speed of groundwater flow and recharge.

c) Ingestion of contaminated soil

Ingestion of contaminated soil risen up by the wind is of course a threat to the health of individuals of all ages, but is more dangerous for children, who are more frequently playing in areas with contaminated soils. This impact pathway is therefore usually analysed just for pre-school children. As hinted above, this pathway is probably more efficient in pollutant transport in an urban context, where pollutants are less likely to infiltrate into the soil. Nevertheless, it can also be significant in rural or semi-rural sites.

d) Dermal contact

Finally, dangerous exposure can occur through dermal contact with toxic substances transported by wind or present on soil. In this case, health risk evaluation should define an average amount of dermal intake of each ATPP, based on pollutant depositions, pollutant soil concentrations, the quantity which can be risen up in a given period, and the average amount of skin surface exposed to ATPP.

Just from this short description of the main micropollutant pathways it is possible to get an idea of the huge amount of specific data required in order to run a full health risk assessment.

Due either to the lack of data indirect health impact assessments usually restrict their evaluations to an Hypothetical Most Exposed Individual (HMEI, or simply MEI). MEI risk

evaluation considers *the worst case* which can be attributed for each different pathway, in order to assess a representative value of the maximum health risk which might be reached.

A typical set of MEI assumptions is e.g. (see Paustenbach et al, 1990) that of a farming family living in the most exposed gridcell of the area considered, harvesting in the same gridcell some vegetables for its own consumption and drinking water taken from the most contaminated surface (or groundwater); possibly also eating contaminated fish caught in the same surface water, and so on. The same kind of hypothesis is also assumed for the other pathways (e.g., children playing almost all day on the worst-case contaminated soil, with the worst-case of dust windblowing, etc.)
Even if MEI assessments can give a plausible (but always to be taken with a grain of salt) estimate of the upper bound of indirect pathways health risk, it clearly appears that they are in any case quite far from reality, and bear little relation to the actual total (and average) *risk* which should be estimated in our approach.

Many very complex mathematical models have been built recently, particularly in the USA, in order to quantify health risk indirect pathways. A few quantifications were made on these aspects, giving us an indication on the order of magnitude of those risks.

Levin (1991), found that *ceteris paribus* (i.e. considering the HMEI for both pathways) indirect pathways contribution to total health risk is in most cases *at least* of the same order of magnitude of the direct one. Sometimes, moreover, they may also be the principal source of risk. This is likely to be the case for dioxins and furans, which are efficiently transported through the food chain reaching breast milk.

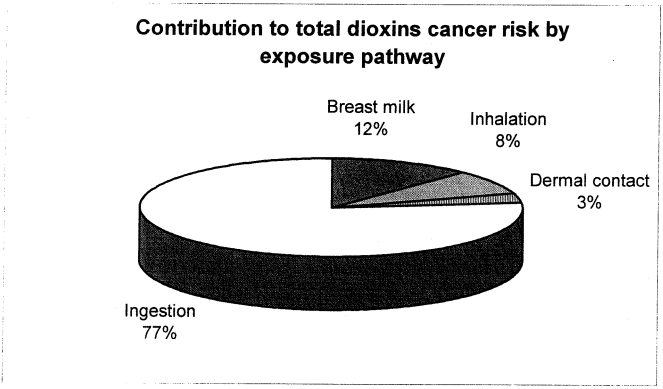



**Figure 3.2** Contribution to total dioxin carcinogenic risk by exposure pathway.
(Source: M. Giugliano -S. Cernuschi, 1991)

Recent research by Giugliano et al. (1994) for a site close to those considered in our study yield even more striking results. The total MEI risk was split into three components which contribute as in Table 3.2

**Table 3.2** Contribution to total carcinogenic risk by exposure pathway

| Pollutant | Damage % by impact pathway | | |
|---|---|---|---|
| | **Inhalation** | **Soil ingestion** | **Dermal contact** |
| Dioxins (I-TEF) | 3 | 95 | 1 |
| Cadmium | 50 | 15 | 34 |

These figures give a substantial idea of the order of magnitude of indirect pathways relative to the direct one. Moreover, since estimates of total mean risk for the *average* individual in the affected area are also provided by the same study and are roughly proportional (for various pollutants and emission levels) to the MEI risk, it may be reasonable to assume that the same ratio between the direct and the two indirect pathways should also hold in our case. A higher estimate will therefore include such pathways and amount to 33 times as much as the base case for pCDDe impacts and twice as much for cadmium. Even with these hypotheses, the mentioned impacts would be considerably less important than those due to macropollutants: These higher estimates should however be considered and particularly transferred to other sites very carefully, as it must be borne in mind that indirect pathways are even more site-specific than inhalation.

## 3.4 Economic valuation

In most cases, as mentioned above, the valuation of health damages was carried out by means of values already used within the ExternE framework (EC DG XII, 1995b), which were only modified to be updated to 1995 prices. The main relevant exception is given by the evaluation of expected additional mortality, that was previously monetised according to the *value of a statistical life* (VSL) - i.e. the value attributed to a change in the probability of death such that one excess death may be expected over a given population. The evaluations carried out in this analysis are rather based on the concept of Years Of Life Lost (YOLL), whose value has been fixed, consistently with the work of other ExternE teams (see EC DG XII, 1997), at 84,330 $ECU_{1995}$. This value was obtained from the VSL used previously, based on given hypotheses on the average number of years lost involved by the loss of a statistical life due to the pollution impacts (see Markandya, 1996; Bickel – Krewitt, 1997), and considering the usual 3% discount rate.

The "original" VSL (3.1 million ECU at 1995 prices) was obtained as an average of several studies based on different methodologies (contingent valuation, hedonic wage, defensive expenditures to reduce lethal risks). While these studies provided a reasonably narrow range of VSL estimates, they are nevertheless affected by several methodological difficulties (Markhandya-Rhodes 1992; Pearce et al. 1992).

Firstly, the valuation was carried out with reference to probability intervals usually much higher than those actually affecting people in our case. The linear extrapolation of WTP estimates to lower probabilities might lead to relevant information losses. In fact, valuation of risk to life and other pathological situations may be affected by nonlinearities as a function of perceived probability which are not accounted for in the ExternE monetisation.

The probability perceived by the laymen exposed to pollution may in turn be sharply different from observed frequencies of adverse events. In particular, empirical studies suggest that public perception tends to overestimate risk with respect to the observed frequencies when these are low (in the order of $10^{-6}$ or lower); conversely, it tends to underestimate risk with respect to the actual frequencies when these are relatively high (see e.g. Viscusi, 1993). The misperception of the risk due to dioxins (and other carcinogenic pollutants) in relation to the "scientifically assessed" probability of getting a cancer may be an enlightening example of this trend.

A few other difficulties not related with the size of the health risk are worth being briefly mentioned:

- the values of the WTP to avoid risk obtained from the empirical studies may be related to voluntariness of risk-taking;
- both the VSL and the value of the YOLL may vary with the age of death according to an unknown relation;
- both the VSL and the value of the YOLL may depend on latency time and circumstances of death, e.g. because of sufferance associated with some diseases.

The role of some determinants of the value of life is not yet clear, and still subject to considerable debate (see e.g. Jones-Lee, 1989; Viscusi, 1993). While we cannot analyse such difficulties in depth, let us just briefly recall some issues concerning relationships between latency times and circumstances of death which are particularly relevant in our case, where carcinogenic pollutants deserved special attention.

We found only one US study (Mitchell and Carson, 1988) reporting a specific (contingent) valuation of willingness to pay (WTP) for reduction of probability of death by cancer *within 30 years* so that one case is spared. Their estimate is equivalent to 105,000 $ECU_{1995}$, about 30 times lower than the VSL used in the ExternE reports (EC DG XII, 1995b). Note that this low value seems to imply undervaluation of actual risk, and is at odds with expectations that cancer deaths be feared more than prevalently old age lethal pathologies attributed to macropollutants.

The monetary values used for cancer risks in this report do not consider the Mitchell-Carson estimate, which was obtained for a specific case (delayed carcinogenic effects from polluted water ingestion) and cannot be easily extended to the kind of risks described in our evaluation. Monetary estimates on cancer risk are rather taken, where possible, from the values defined within the ExternE project (EC DG XII, 1997). These values were obtained based on the YOLLs, plus an estimated specific cost of illness. It has to be stressed, however, that further research would be necessary on this issue to reach more precise estimates. The value for

cancers induced by dioxins was set equal to the highest value among carcinogenic effects (i.e., 2.06 million ECU).

The values ultimately used to estimate external costs are those listed in Table 3.3.

**Table 3.3** Monetary values utilized to assess the external costs

| Event | Value ($ECU_{1995}$/case) |
|---|---|
| Broncodilator usage | 37 |
| Cough days | 7 |
| Lower respiratory symptoms (wheeze) | 7.5 |
| Congestive hearth failure (elderly) | 7,870 |
| Case of chronic bronchitis (children) | 225 |
| Chronic bronchitis (adults) | 105,000 |
| Chronic cough (children) | 225 |
| Restricted Activity Days (RAD) | 75 |
| Respiratory Hospital Admission (RHA) | 7,870 |
| Cerebrovascular Hospital Admission | 7,870 |
| Emergency Room Visits (ERV) for Chronic Obstructive Pulmonary Disease | 223 |
| Symptom Days | 7.5 |
| Mortality ("chronic" YOLL) | 84,330 |
| Mortality (VSL) | 3,100,000 |
| Leukemia (fatal) | 2,060,000 |
| Leukemia (non-fatal) | 450,000 |
| Lung cancer (fatal) | 1,330,000 |
| Lung cancer (non-fatal) | 450,000 |
| Nasal cancer (fatal) | 1,100,000 |
| Nasal cancer (non-fatal) | 450,000 |
| Stomach cancer (fatal) | 1,240,000 |
| Stomach cancer (non-fatal) | 450,000 |
| Occupational injuries (temporary) | 77.5 |
| Occupational injuries (permanent) | 23,700 |

**Table 3.4** Global warming and ozone valuation units

| Pollutant | Monetary unit value (ECU/t pollutant) |
|---|---|
| Global Warming | |
| $CO_2$ | 18 - 46 |
| $CH_4$ | 400 - 520 |
| Ozone precursors | |
| NOx – total | 1494 |
| *of which:* | |
| *Mortality* | *412* |
| *Morbidity* | *723* |
| *Crops* | *350* |
| VOC – total | 134 |
| *of which:* | |
| *Mortality* | *39* |
| *Morbidity* | *71* |
| *Crops* | *24* |

## 3.5 Estimated impacts and damages of waste incineration

The external costs of energy recovery from the incinerator are reported separately (i.e. before computing the impact *net* of the landfill external cost) in terms of mECU per kWh recovered (Table 3.9). Although this reporting format gives some useful information, and makes a comparison possible with more traditional fuel cycles analysed within the ExternE programme, we do not consider these results as the most significant to evaluate the fuel cycle as we have defined it in section 1.2.1.

A few results illustrated in the tables below are worth being underlined and commented. Leaving aside for a moment the damage caused by global warming (that will be commented later), results show that public health, and in particular mortality, explains most of the total damage for the incinerator case. The same does not hold for the landfill case, where the "amenity impact" accounts for about 66% of the total (greenhouse gases excluded).

The damage caused to agriculture and forests (even when including the effects of ozone) is of second order as compared to public health. The same applies to material damages.

Going back to public health damage, the results confirm what was found in the previous study (IEFE 1995) as regards the damage due to micropollutants (heavy metals and dioxins): their external cost, computed in mECU/kWh or in ECU per tonne of waste disposed, proves to be negligible as compared to the impact caused by primary and secondary macropollutants. This outcome is obviously no longer valid when we look at it in terms of external cost per tonne of *pollutant emitted*: on this respect dioxins, heavy metals and most carcinogenic micropollutants show a damage considerably higher (*ceteris paribus*) than any macropollutant, as from Table 3.5 and Table 3.6.

**Table 3.5** Damage per tonne of macropollutant emitted (in ECU/t)

| **Stage** | **Pollutant damage (ECU/t)** | | |
|---|---|---|---|
| | **PM** | **NOx (nitrates)** | **SOx (sulphates)** |
| Incinerator power gen. | 50,506* | 13,244 | 10,173 |
| Transport | 3,974,280** | nq | nq |

** Here the incinerator damage refers to $PM_{10}$, while transport damage refers to $PM_{2.5}$, whose ERFs for all impact categories are estimated to be 1.67 times higher than the corresponding $PM_{10}$ ERFs. Moreover, it is worth recalling that transport particulates are emitted close to the ground and their dispersion has been estimated with a software different from Ecosense 2.0.

**Table 3.6** Damage per tonne of micropollutant emitted (in ECU/t)

| **Stage** | **Pollutant damage (ECU/t)** | |
|---|---|---|
| | **Cadmium**** | **Dioxins** (TCDDe)* |
| Power generation | 345,000,000 | 1,097,338,000 |

* The value of the external cost due to dioxins was estimated based on the value of 2E-07 cancer cases per GWh, obtained in IEFE (1995). The software Ecosense was not sensitive enough to treat such a low emission and damage levels.

**Table 3.7** Damages of energy recovery from waste incineration (facility of Milano – Silla)

| | mECU/kWh | $\sigma_g$ |
|---|---|---|
| **POWER GENERATION** | | |
| Public health | 26.6 | |
| Mortality*- YOLL *(VSL)* | 22.5 (80.6) | B |
| *of which TSP* | *0.9 (3.4)* | |
| *$SO_2$* | *3.4 (12.5)* | |
| $NO_x$ | *17.6 (64.7)* | |
| $NO_x$ *(via ozone)* | *0.66* | |
| Morbidity | 4.1 | |
| *of which TSP, $SO_2$, $NO_x$* | *2.9* | A |
| $NO_x$ *(via ozone)* | *1.17* | B |
| *Hg + Pb* | ng | |
| Cancer | | |
| *Cd + pCDD* | 0.026 | B |
| Accidents | ng | A |
| Occupational health | nq | A |
| Major accidents | - | - |
| Crops | nq | B |
| *of which $SO_2$* | *0.002* | |
| $NO_x$ *(via ozone)* | *0.56* | |
| Ecosystems | ng | B |
| Materials | 0.25 | B |
| *Monuments* | nq | |
| Noise | nq | |
| Amenity losses | nq | |
| Global warming | | C |
| *(Illustrative restricted range)* | | |
| low | 22.3 | |
| upper | 57.0 | |
| **OTHER FUEL CYCLE STAGES** | | |
| Public health | 1.65 | A-B |
| *Outside EU* | - | |
| *Inside EU* | *1.65* | |
| Road accidents | 0.25 | A |
| Occupational health | 0.94 | A |
| *Outside EU* | *nq* | |
| *Inside EU* | *0.94* | |
| Ecological effects | ng | B |
| Road damages | ng | A |
| Global warming | | C |
| *(Illustrative restricted range)* | | |
| low | 0.013 | |
| mid | 0.033 | |

*Yoll= mortality impacts based on 'years of life lost' approach, VSL= impacts evaluated based on 'value of statistical life' approach.
ng: negligible; nq: not quantified; iq: only impact quantified; - : not relevant
Numbers in brackets are computed by using the Value of Statistical life instead of the YOLL.

In addition to that, both tables show how relevant the "transport" stage can be for the fuel cycles analysed. The relevance of this stage will remain considerable even when considering the *differential* impact between the two waste treatment technologies (see Table 3.13). As shown in Table 3.5, this is more properly underlined when looking at the extraordinarily high data obtained *per unit of pollutant emitted.* According to the results recently found by the ExternE – Transport project (EC DG XII, 1997), the vast majority of the impact coming from transports falls in a local area – where "local", as mentioned before, is referred in this case to a smaller region than the traditional 100 x 100 squared km considered in Ecosense. This means that the results shown below for transport externalities are particularly site-specific, and are heavily influenced by the high population density (indeed, one of the highest in Europe) of the area where waste transport occurs. The *local* effects of sulphates and nitrates were not quantified in the ExternE project mentioned, since this analysis requires particularly complex analyses. The regional damage only, as quantified by Ecosense for these secondary pollutants, accounts for less than 10% of the overall damage.

## 3.6 The evaluation of disamenities of waste disposal site

### 3.6.1 The social impact of solid waste landfills and incinerators

Solid waste landfills and incinerators have been one of the most frequent targets of public opposition within the industrial world over the last decades so that an increasing scientific and economic research effort has been devoted to the analysis and evaluation of their environmental impact. The main effects usually considered for landfills include:

- the risk of soil and groundwater pollution by landfill leachate, despite isolation practices used in modern plants;
- generation of biogas, particularly where municipal and other organic waste is dumped. While in the past the main perceived risk of such gas was related to blasts and fires due to its large methane component this has been sharply reduced in moderns plants by collection and controlled burning of most of the gas. However, the uncollected fraction (typically about 40%) contains trace components which are the source of annoyance due to their odours (hydrogen sulphide, mercaptans) and may cause health risks (benzene and other aromatics) or damage to agriculture;
- visual impact of the landfill site during cultivation, although this may be eventually beneficial if it provides restoration of already existing holes, as in the case of abandoned quarries;
- dissemination of pathogenic micro-organisms by mice, birds and other animals feeding on the site;
- contribution of biogas to the global warming problem is regarded as serious by environmental experts but hardly a main concern of local opponents.

Turning to incineration, its main impact lies with air pollution by particulates (including their heavy metals component), sulphur and nitrogen oxides, and organic chlorinated compounds. These components may be responsible of various damages to public health, agriculture and construction materials. Carbon dioxide emissions also contribute to global warming. Moreover, bottom ashes and other solid residuals of the incineration process must be landfilled anyway, so that the previously discussed items also apply to some extent.

The visual impact of the plant should also be considered, while other disamenities are common to landfilling, incineration and other waste treatment practices: these are truck traffic carrying the refuse to the plant and the impact of temporary storage, which is probably conceptually similar to that of the landfill although it regards far smaller amounts. It must be added that impacts are often boosted by the stigma often attached to areas where such plants are located

### 3.6.2 Existing valuation studies

Environmental economists have faced the valuation problem of such impacts in two fundamental ways. Wherever it is feasible to trace back damages to pollutant emissions or leakages by modelling their soil or atmospheric dispersion and applying suitable dose-response relationships, the monetary valuation could be applied to physical damage caused by the technology. This approach has been followed in other sections of this chapter. A simpler variant of this approach requires that, under suitable assumptions, monetary valuation be applied directly to emission rates, as in the study promoted by UKDOE (1993).

However, it is clear that in the case of incinerators, and even more of landfills, a large share of the impact is either inherently subjective (as in the case of odours and visual impact, and to some extent of traffic) or very difficult to work out by scientific means (as for the risks of groundwater pollution and pathogenic dissemination or social stigma effects), so that welfare loss due to technology is probably more related to perceptions than to science-based impact assessment. The stochastic nature of some of the most feared events and of their consequences (notably for human health) add further uncertainties to the valuation process as risk aversion of potentially affected parties should be taken into account.

Therefore, it is not surprising that several researchers have tried to evaluate the global impact of waste disposal sites by following an alternative, holistic approach that amounts to direct evaluation of utility losses from the site, by contingent valuation or as reflected by models of real estate prices through hedonic techniques. UKDOE (1993) provides a survey of studies performed in North America while no study to our knowledge has been published concerning European countries. Hedonic pricing has been the preferred methodology, perhaps due to the deep emotional impact raised by solid waste disposal sites, or to largely unknown burden of plants, which may affect contingent valuation. The present study follows this prevailing stream.

The main drawback of the holistic approach lies in that, while it can in principle lead to a complete cost-benefit analysis of technology choice and siting decisions, it does not allow to identify any impact pathway or (in other words) to disentangle any reason for the public opposition to the plant. Thus, while this approach can provide estimation of external costs which may be used for the implementation of policy instruments like levies or compensations to affected parties, it cannot be used to generate guidelines for the improvement of disposal practices. For example, it cannot tell us whether more effort should be devoted to the abatement of atmospheric emissions rather than to curbing subjectively perceived disamenities.

Hedonic pricing studies have not always detected external costs of landfills, and none has estimated those of incinerators. In some cases, where the exploitation of isolated sites has triggered infrastructure development, the impact on house prices has been reported as positive (see Nelson et al.., 1992). Some of these mixed results are surveyed by Michaels and Smith (1990), who developed a strategy based on a careful modelling of announcement effects of hazardous waste sites and on more accurate market segmentation. The role of information concerning pollution risks has also been the focus of research by Kohlhase (1991) and Mendelsohn et al. (1992). This is certainly a sensible strategy for the valuation of impacts of hazardous waste sites, where the main impact pathway is the feared contamination of the groundwater table.

In all cited studies, the site impact on house prices is measured by distance of the affected property from the site. Again, this is an acceptable proxy for visual impacts and also for exposition to pollutants if their actual dispersion pattern are not known. However, in the case of municipal solid waste, we suspect that a significant impact can be attributed to odours, which are themselves a source of annoyance and may reveal even to lay people the presence of other, nondetectable noxious gases. Note that noxious components of landfill and incineration gases are not often monitored by air quality control services.

This fact has been partly acknowledged by Hwang (1979) who found that prices in the Chicago metropolitan area were positively affected not only by distance from a sanitary landfill but also by degrees away from prevailing wind over the site. The present study follows this line by constructing a more accurate measure of odour exposition, in order to check for the hypothesis that this factor is crucial in the explanation of externality patterns around a waste disposal site.

### 3.6.3 An hedonic pricing study

This work follows the hedonic pricing technique as described e.g. by Palmquist (1991), Bateman (1993) or Freeman (1993). The study was implemented in a suburban area north-west the town of Milan, comprising 16 municipalities with 268,700 inhabitants and a mean population density of 1648 per squared kilometre. The analysis focused in particular on the landfill of Cerro Maggiore; some conclusions were also found for the incineration plant located in the SW part of Busto Arsizio, (about 9 km west of the landfill).

No general reliable databases of house market transactions exist in Italy. A sample of 289 sales occurring between 1993 and 1995 was therefore built thanks to the collaboration of Tecnocasa, a major real estate agency which operates a large network of realtors. These operators supplied not only structural data about apartments but also their own quality judgements about buildings and the inside condition of dwellings. Moreover, they could offer information about buyers which will be exploited for the estimation of marginal willingness to pay for various characteristics (not included in this paper).

The dependent variable selected for the model price in million liras per square meter rather than home price: this helps to curb heteroskedasticity problems and allows a clearer identification of contribution to price of each characteristic. As Parsons (1990) showed, regression of total price on characteristics is biased unless neighbourhood variables are weighted by lot size. Our specification achieves much the same goal.

The meaning of structural independent variables is as follows:
MQ: size of the dwelling in squared meter, assessed by commercial criteria (sample mean is 90.7)
FLOOR: floor of flat; 0.5 if both ground and first
NOLIFT: floor of flat if greater than first and without lift (14% of sample)
BATH. number of bathrooms, weighted by MQ
GARAGE: covered private car park: 2 if included in home price, 1 if not included, 0 if none
GARDEN: 2 if private, 1 if common, 0 if none
REFURB. dummy variable for recent refurbishment (28.7% of sample)
JUDG: sum of realtor's evaluation of the building and dwelling condition, expressed as 1 if fair, 2 if good, 3 if excellent.
AGE: age of the building at time of sale (years)
DETACH: dummy variable for detached or semidetached house (9.7% of sample)
TERR. dummy variable for terraced house (7.6% of sample)
PORTER: dummy variable: 1 if the block has a porter (2.4% of sample)

Other variables that were tried but not found significant include: number of balconies, terrace, independent heating number of rooms, class of housing, road exposition, southern exposition, uncovered car park, cellar, availability (if rented or free), impending mortgage. Prices in the area are reported as quite stable across the sampling period and regression analysis could not detect any time pattern of prices.

Neighbourhood variables could be collected at municipality level, except distances that were evaluated in km for each home:
INDPC: industrial employment per 1000 inhabitants, as a proxy of industrialisation of the municipal area
DENS: population density (inhab./sqkm), as a proxy of "residential nature" of the municipality
SOCPC: employees of health and social services per 1000 inhabitants
MILAN: linear distance from the main business centre, the city of Milan.

Among non-significant variables that were considered but dropped from the model were: employees of education and of other services, distances from: local parks, the nearest railway station, main business centres within the area and from the only university located in the area, local property tax rates.

In this stage of the analysis, environmental amenities were measured by simple linear distance from the Cerro landfill site (LFILL) and from the Busto incinerator (INC).

Regression results are presented in Table 3.8 for a linear and semilog functional form. A linear Box-Cox flexible form yielded almost the same results as the semilog, suggesting that this is the most appropriate specification. The double-log form was clearly worse.

All coefficients of both specifications have the expected sign and relative sizes appear to be consistent. The negative sign of the dwelling size (MQ) coefficient shows that smaller homes are relatively more expensive, so that home prices are a quadratic function of size. The PORTER variable should be interpreted as a proxy for larger blocks of flats where this kind of service is more likely found in a suburban area, but are probably regarded in general as a poorer type of housing, hence the negative sign.

However, both environmental variables are not significant and LFILL also has the wrong sign. This is not very surprising in the case of the incinerator, due to its relatively small dimensions, its location among woods outside residential areas and the long habit of neighbouring population to live nearby. On the other hand, in the case of the landfill we suspected that the main perceived disamenity was the odour coming from the plant and that is reported as very annoying under the most stable atmospheric conditions. Therefore, an indicator of people's exposition to this annoyance was constructed instead of linear distances.

**Table 3.8** Estimation results when environmental amenities are measured by linear distance from the landfill and incinerator sites

| | Linear model | | Semilog model | |
|---|---|---|---|---|
| **Variable** | Coefficient | T-Statistic | Coefficient | T-Statistic |
| C | 2.39090 | 5.12 | 0.75572 | 3.41 |
| MQ | -0.00583 | -5.91 | -0.00294 | -7.10 |
| FLOOR | 0.05178 | 2.04 | 0.02311 | 2.23 |
| NOLIFT | -0.05823 | -2.29 | -0.01785 | -1.35 |
| BATH | 0.27516 | 4.61 | 0.13603 | 4.90 |
| GARAGE | 0.10600 | 3.89 | 0.06209 | 4.50 |
| GARDEN | 0.05413 | 1.84 | 0.02967 | 2.11 |
| REFURB | 0.05118 | 1.01 | 0.06457 | 2.38 |
| JUDG | 0.14633 | 6.25 | 0.07776 | 6.82 |
| AGE | -0.00501 | -5.57 | -0.00329 | -6.89 |
| DETACH | 0.91918 | 6.73 | 0.44601 | 8.14 |
| TERR | 0.30715 | 2.91 | 0.14721 | 3.43 |
| PORTER | -0.50241 | -2.84 | -0.22964 | -2.51 |
| DENS | -0.00019 | -1.95 | -0.00009 | -1.79 |
| INDPC | -0.00057 | -1.37 | -0.00040 | -1.91 |
| SOCPC | 0.00894 | 2.79 | 0.00491 | 2.96 |
| MILAN | -0.02845 | -3.71 | -0.01345 | -3.51 |
| LFILL | -0.00828 | -0.48 | -0.00395 | -0.44 |
| INC | 0.00365 | 0.24 | 0.00710 | 0.93 |
| $R^2$ | 0.676 | | 0.712 | |
| Adjusted $R^2$ | 0.654 | | 0.693 | |
| S.E. of regression | 0.369 | | 0.183 | |
| Sum squared resid. | 36.802 | | 9.014 | |
| Log likelihood | -112.277 | | 90.996 | |
| F-statistic | 31.263 | | 37.132 | |

### 3.6.4 An index of landfill annoyance

Ideally, in order to test the impact of exposure to odours on property prices, actual measurements of perceived effects should be used. However, monitoring network do not exist for gases which may represent tracer indices of odours: for our site, these were only monitored at the landfill site as a component of total landfill gas. Therefore, we replaced actual measurements by values estimated by means of a dispersion model.

The ground level concentration distribution due to the landfill emissions has been evaluated using the USEPA's Industrial Source Complex Short Term (ISCST2) (U.S. E.P.A., 1992). The ISCST2 model is a steady-state Gaussian plume model which can be used to assess pollutant concentrations from a wide variety of sources associated with an industrial source complex. This model can account for settling and dry deposition of particulate, downwash effects around buildings, emissions from point, area, line and volume sources, plume rise as a function of downwind distance and limited terrain adjustment. ISCST2 is designed to calculate the average ground level concentration or total deposition from multiple continuous point, volume and/or area sources in 1, 2, 3, 4, 6, 8, 12 and/or 24-hour time periods. Provision is made for special discrete (x,y) receptor points that may correspond to sampler sites, points of maxima, or special points of interest. Sources can be positioned anywhere relative to the grid system.

The ground level concentration of a pollutant C(x,y) is calculated using the following equation (1):

$$C(x,y) = \frac{Q}{2u\pi\sigma_y\sigma_z} \exp\left[-0{,}5\left(y/\sigma_y\right)^2\right]\left\{\exp\left[-0{,}5\left(H/\sigma_z\right)^2\right] + \sum_{i=1}^{\infty}\left\{\exp\left[-0{,}5\left(H+2iH_m/\sigma_z\right)^2\right] + \exp\left[-0{,}5\left(-H+2iH_m/\sigma_z\right)^2\right]\right\}\right\}$$

where:

C = ground level concentration [mg/m$^3$]
Q = pollutant mass flow [g/s]
u = wind speed at stack height [m/s]
$\sigma_y$ = dispersion coefficient of horizontal distribution of concentration [m]
$\sigma_z$ = dispersion coefficient of vertical distribution of concentration [m]
H = effective height of emission [m]
$H_m$ = mixing layer height [m]
x = downwind distance from the source [m]
y = crosswind distance from the source [m]

Dispersion coefficients $\sigma_y$, $\sigma_z$ depend from atmospheric turbulence and downwind distance from the source and are calculated by Pasquill-Gifford's formulations. Effective emission height results from the sum of stack height and plume rise, evaluated through Brigg's formulation as a function of stack gas speed, temperature and atmospheric stability in terms of

Pasquill's categories. Trapping conditions arising from the diffusion in a vertical nonhomogeneous atmosphere with a stable layer limiting the vertical dispersion of the pollutant emitted in the underlying neutral or unstable layer, are accounted for in the model through the multiple reflections method, which considers ground level and the base of the atmospheric stable layer as perfect reflectant surfaces. Second member summatory in equation (1) is the mathematical expression of this method. This procedure is applied only in case of unstable or neutral Pasquill's categories.

The landfill has been regarded as a ground level area source, with an estimated biogas emission of 2000 $m^3/h$. A Cartesian grid was defined to locate emission source and receptors: x-axis is in the East-West direction while y-axis is in the North-South direction. The position of the landfill was taken as the origin of the Cartesian axes. Both co-ordinates range between -8 and 8 km, with a step of 1 km. Flat terrain option has been utilised.

Odour diffusion has been evaluated by considering hydrogen sulphide ($H_2S$) as a tracer of malodorous substances emitted from the landfill. This choice is justified by the fact that $H_2S$ is the compound with the lowest value of odour threshold (Bertoni D. et al., 1993) among detected biogas components. In a conservative approach, $H_2S$ concentration in biogas was considered as the maximum recorded concentration during biogas composition analyses (55 $mg/m^3$), with a resulting emission rate of 30 μg/s.

After a particularly unfavourable combination of wind speed and atmospheric stability had been identified (wind speed 1 [m/s], Pasquill's category 'F), contour values of hourly concentrations have been determined for all wind directions. Because of the particular shape of the landfill, circumferences centred on the landfill site were obtained. On the basis of the joint frequency of wind speed, wind direction and atmospheric stability category corresponding to the critical meteorological combination considered, the number of hours per year in which this combination effectively occurs has been determined and reported in Fig.3.3 for each octant of the wind rose. The contour values of odour threshold for H2S (0.7 $\mu g/m^3$) are also represented as a dashed line in Fig.3.3. All data utilised in the evaluation are from the nearest meteorological station to the landfill site (Milano Malpensa), which can be considered well representative of the local climate at the emission site.

Ground level concentrations as a function of the distance from the landfill site can be expressed through the following equation:

$$C = 3.66 \exp(-0.22\, r) \qquad (3)$$

where:

C = ground level concentration [$mg/m^3$]
r = distance from the landfill [km]

The former, owing to the symmetry of the emission source, can be applied for every direction.

In order to properly assess the impact of odorous emissions from the landfill site both intensity and duration of exposure must be taken into account; thus, an "odour exposure index" (EI), given by the ratio between the ground level concentration and the odour threshold of $H_2S$ times the annual number of hours in which the meteorological condition considered occurs, has been defined for each octant of the wind rose. No impact is expected where the ground level concentration is lower than the odour threshold. Since the odour threshold is positioned at 7.5 km from the landfill site, the expression of the exposure index of a dwelling i as a function of distance from the landfill site in the j-th octant is therefore given by the following equations:

$$EI_{ji} = N_j\, 5.23 \exp(-0.22\, r_i) \quad \text{for } r_i \leq 7.5 \text{ km} \qquad (3')$$
$$EI_{ji} = 0 \quad \text{for } r_i > 7.5 \text{ km}$$

where:

$EI_{ji}$ = exposure index in the j-th octant [h]

$r_i$ = distance from the landfill of dwelling i [km]

$N_j$ = annual number of hours of occurrence of the meteorological combination considered in the j-th octant.

### 3.6.5 Integrating dispersion modelling results into the hedonic model

The exposure index EI estimated in the previous section was evaluated for all locations of dwellings of our sample. This is easily done by using the LFILL variable for the distance in eq. (3) and by assigning each dwelling of the sample to its octant.

The individual EI value (divided by 1000) was then substituted in the hedonic equation estimation for the LFILL variable. Results are given in Table 3.9. Distance from the incinerator remained not significant and was dropped from the model.

Note that the new EI variable is now significant at 10% in the linear specification and at 5% in the semilog, and has the expected sign. Again, the semilog form is preferred both on statistical grounds (as the linear Box-Cox transformation yields very similar results) and for theoretical reasons as it is consistent with a decreasing marginal effect of the disamenity.

The existence of the relationship as found in the data is also reinforced by the fact that 43% of buyers within our sample come from the same town and another 42% from neighbouring towns, therefore sufficient information should be available concerning environmental disamenities of properties, not less than for other neighbourhood variables of the model.

For example, from the estimated equations it can be computed that for an exposure index of 1000 (i.e. 1000 hours a year at the threshold level or 500 hours at twice that level)), the mean property value loss amounts to 4.1% in the linear specification and to 5.1% in the semilog, while for an exposure of 2000 (typical of the town of Cerro Maggiore located just south of the landfill site) the loss is raised to 8.2% in the linear and 9.9% in the semilog.
The hypothesis that perceived impact is related to odours is therefore supported. Generalisation of results to other sites or to other similar sources of annoyance should

however be carried out cautiously. While the perceived impact is likely to be *related* to odours, its effect may not entirely *depend* on this kind of annoyance, which may be just turning the buyer's mind to other effects, e.g. contamination of drinking water.

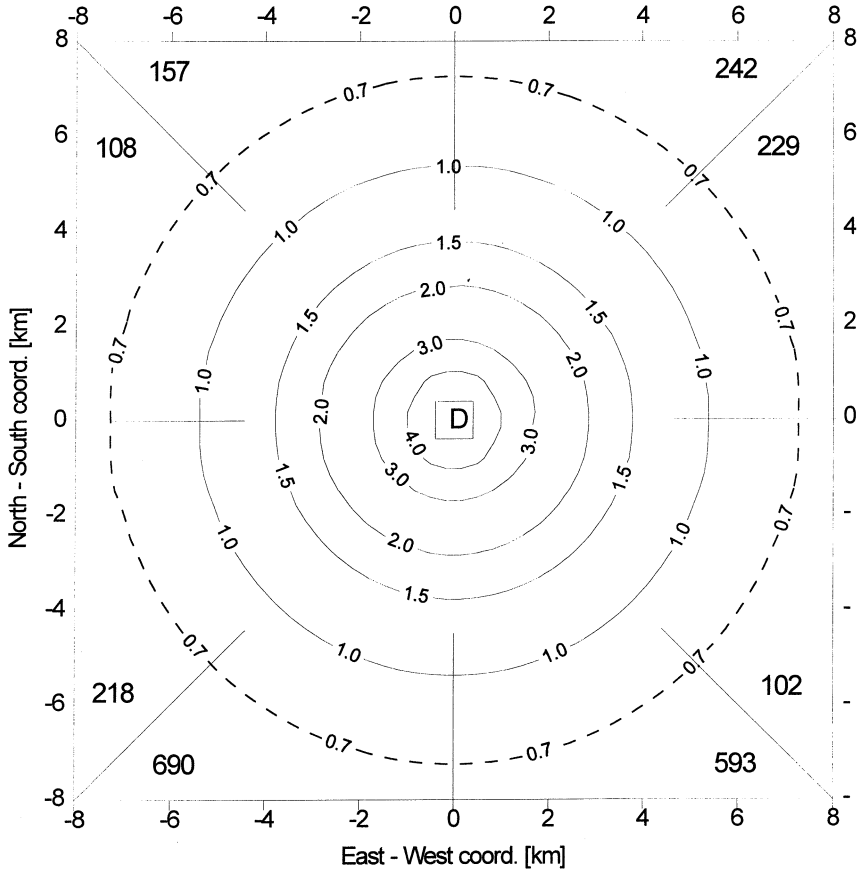



**Figure 3.3** Contour lines of ground level concentration of $H_2S$ [µg/m³] for the combination of wind speed 1 [m/s] and atmospheric stability F. In each octant the number of hours per year in which this combination effectively occurs, is reported.

### 3.6.6 External cost estimation and open issues

The evaluation of the external costs of a facility like a large sanitary landfill can be carried out starting from the hedonic pricing model, as suggested by environmental economics handbooks. In our study, data about property buyers' socio-economic characteristics were also collected so that it should in principle be feasible to estimate a function representing marginal willingness-to-pay to avoid exposure to landfill disamenities. This implicit price should then be applied to actual properties of the affected area.

This exercise lies however beyond the scope of this paper. The aim of this section is to develop a preliminary estimation under simplifying assumptions in order to give an idea of orders of magnitude as well as of difficulties of applying standard methodology, mainly developed for the urban clean air problem or for the remediation of hazardous waste sites, to this rather different kind of externalities.

A linear hedonic equation involves a constant marginal willingness to pay for environmental quality. While this is in general theoretically dubious, it has been claimed that this assumption may be acceptable in two special case: if households have similar utility functions and socio-economic conditions, or (for different reasons) if the affected area experiencing variation of the amenity within the sample is a minor part thereof (Freeman, 1993: 397). While over 86% of the sample experienced positive values of the EI exposure index, only 40% were affected by values above 500, which involves a price effect larger than 2%. The second special case just quoted may therefore offer some justification for the use of this simplified approach.

Let us therefore assume that marginal effect of the landfill as measured by EI is correctly estimated by the linear specification of Table 3.9 and denote it by W.

The affected area is approximated as a circle centred around the landfill site, with a radius of 7.5 km (the maximum distance of the odour perceptive threshold in the worst climatic condition, as from sec. 4). However, a smaller circle of 1 Km radius is supposed to be population-free and does not contribute to damage.
Total damage per squared meter of dwelling area is given by:

$$D = \frac{1}{8}\Sigma_j N_j \cdot V \cdot W \qquad (4)$$

where Nj values are found in Fig.3.3 and V represents total exposure of the affected area as measured by EI which can be calculated as

$$V = 2\pi \int_1^{7.5} rae^{br}\, dr \qquad (5)$$

$$= 2\pi ab^{-2}\left[e^{br}(br-1)\right]_1^{7.5}$$

where a and b are given by eq. (3'). The integral is cut at the level where EI falls below the perceptive threshold and at 1 km from the landfill site.
Assuming further a uniform population and housing density in the affected area equal to the actual values of the sample area, i.e. 1648 inhabitants per squared kilometre and 25 square meters per capita, we obtain:

D= 303.16 billion liras.

While the above formula (5) is more suitable for implementation of the model in various situations (provided that results could be exported), one could simply use the regression equation of table 3.9 and forecast the average price change due to the landfill, by assuming that EI equal zero throughout the sample. This exercise has the advantage of a better approximation to actual damage to the extent that the sample is representative. In our case, an average price change of 2.8% is estimated. If extrapolated to the population with the characteristics just described, the damage amounts to 397 billion liras. This higher estimates is not used in the following.

For policy purposes, it would be useful to relate this value to a measure of activity, that is municipal solid wastes dumped at the site[i]. This final passage raises two issues.

The first issue is related to landfill gas and odour emissions. It is well known to environmental experts that these effluents follow a deferred time pattern with respect to waste disposal. In particular, the delay of odour emissions could be longer than that of other biogas components, as noted in our case study, for largely unknown reasons, but this further delay is ignored. Therefore, the previous valuation has been corrected to account for the delay of emissions (and hence damage) with respect to waste dumping. The use of a 3% discount rate effectively amounts to multiplying the above values by 0.825.

Second, the estimated damage is a stock (real estate value) measure which cannot be directly compared with the benefits of a flow of solid waste disposed of at the site. The easiest way out is to compare total external damage with total waste disposal at the site, but this requires heroic assumptions on the households' expectations concerning future activity at the site at the time they paid an implicit price for the disamenity. Since a landfill can be closed down and its impact sharply decrease (more than in the case of urban air quality), such assumption would dramatically affect the estimation of external costs per unit of dumped refuse.

It seems more sensible to assume that the estimated stock damage represent the capitalisation of the current disamenity flow cost, to be compared with current private benefits of the activity. To follow this way, damage cost must be turned into an yearly flow. By applying a rent/value ratio of 0.06 and considering that the site under scrutiny has been receiving an average 2,000 tonnes/day in the three years, we obtain a mean external cost of 20,842 liras per tonne of solid waste (i.e. about 13.2 ECU/t).

**Table 3.9** Estimation results when the odour exposure index (EI) is introduced in the model

| | **Linear model** | | **Semilog model** | |
|---|---|---|---|---|
| **Variable** | Coefficient | T-Statistic | Coefficient | T-Statistic |
| C | 2.75304 | 9.14 | 1.11482 | 7.29 |
| MQ | -0.00587 | -5.84 | -0.00295 | -7.03 |
| FLOOR | 0.05346 | 2.11 | 0.02441 | 2.38 |
| NOLIFT | -0.06348 | -2.48 | -0.02099 | -1.59 |
| BATH | 0.27109 | 4.59 | 0.13299 | 4.78 |
| GARAGE | 0.10567 | 3.91 | 0.06186 | 4.49 |
| GARDEN | 0.06236 | 2.07 | 0.03475 | 2.37 |
| REFURB | 0.05516 | 1.10 | 0.06740 | 2.50 |
| JUDG | 0.14831 | 6.32 | 0.07858 | 6.93 |
| AGE | -0.00473 | -5.10 | -0.00312 | -6.39 |
| DETACH | 0.91033 | 6.60 | 0.43854 | 7.87 |
| TERR | 0.30743 | 2.95 | 0.14684 | 3.53 |
| PORTER | -0.44575 | -2.87 | -0.19290 | -2.44 |
| DENS | -0.00019 | -2.51 | -0.00011 | -2.70 |
| INDPC | -0.00082 | -1.93 | -0.00059 | -2.73 |
| SOCPC | 0.00743 | 2.19 | 0.00415 | 2.52 |
| MILAN | -0.03742 | -6.74 | -0.02072 | -6.72 |
| EI | -0.07885 | -1.74 | -0.05187 | -2.07 |
| R-squared | 0.677 | | 0.713 | |
| Adjusted R-squared | 0.657 | | 0.695 | |
| S.E. of regression | 0.368 | | 0.182 | |
| Sum squared resid. | 36.636 | | 8.995 | |
| Log likelihood | -111.624 | | 91.308 | |
| F-statistic | 35.487 | | 39.581 | |

If such waste were incinerated rather than landfilled, by assumptions of this study it would be possible to obtain a net electricity production of 888 kWh/tonne, which reduces to a net gain of 840 kWh/tonne if we account for the fact that energy recovery from burning methane contained in recovered landfill gas would be foregone. However, externalities arising through the landfill odours would also disappear, even though some disamenities and other external effects may still be entailed by the (far smaller and odour-free) landfill site required for incinerator bottom ashes. Thus, if incineration of solid wastes is to be regarded as an electricity production cycle, it would *avoid* the landfill disamenities analysed in this paper at the rate of 24.8 liras/kWh (about 12.95 mECU/kWh), to be subtracted from other incineration externalities.

Further research effort should therefore comprise:

- development and comparison of a different exposure index, consisting of the number of hours in which odours are above the perceptive threshold in each receptor;
- definition of a landfill emission life cycle, as already done in IEFE (1995) for the estimation of global warming impacts, in order to single out a more reliable relationship between waste disposal, effluent emissions, and impacts;
- inclusion of several measures of damage from a given marginal WTP to get rid of the plant, as suggested by Bateman (1993);
- sensitivity analysis of the relationship between stock value losses and annual external cost, also by allowing for the presence of estate value taxation, as requested by Freeman (1993).

On the other hand, if we interpret our results by assuming that perception of the externalities are related to odours but the estimated willingness to pay does actually include the implicit price of other landfill disamenities, then the estimated value should be regarded as a minimum which is not far away from actual external cost of dumping solid waste at the site.

In any case, it is worth recalling that this value is at most related to incurred perceived disamenities. External costs suffered by other parties than the residents of the considered area are not included: this is the case of other productive and commercial activities as well as of groundwater pollution suffered by residents farther away downstream of the site. The impact of this plant on long range air pollution is estimated in section 3.9 and its contribution to global warming in section 3.8.

## 3.7 Public health impacts from accidents

The estimation of the occupational health damage for both technologies was based on country-specific data.

### 3.7.1 Incineration plant building

We proceeded by estimating an average coefficient in terms of number of accidents per billion $ECU_{1995}$ spent in the construction industry. The data we found for Italy yield the following average ratios:

5.7 accidental *death* cases per billion $ECU_{1995}$;
123.1 *permanent damages* to workers per billion $ECU_{1995}$;
1,892 *temporary damages* to workers per billion $ECU_{1995}$;

where temporary damages are defined as any labour accident causing more than 3 days absence from work, but with a following complete recovery; permanent damage is defined as any accident which brings about any kind of irreversible invalidity. Data on construction accidents were taken from INAIL Statistics.

The average amount required to build an incinerator plant of 900 t/d capacity is about 110 million $ECU_{1995}$ (MECU). On the other hand, this plant would save the construction of a sanitary landfill of the same capacity, which would cost about 25 MECU. It has to be stressed, however, that the available Italian data on the investment costs necessary to realise landfills are poorer as compared to the incinerator case. The available sources show quite a wide range of numbers, varying from 12 up to more than 100 millions ECU for a landfill of the size here described (see e.g. Kaulard-Massarutto, 1997 and CUEIM-IEFE 1997). Moreover, the incinerator would not completely replace the landfill site, as it would still require a smaller disposal facility for bottom ashes, slag and exhausted pollution prevention.

**Table 3.10** Public health accidents impact for incinerator and landfill construction

| | **Incinerator** | **Landfill** |
|---|---|---|
| Size (capacity, in t/day)) | 900 | 2000 |
| Exp. death cases | 0.64 | 0.15 |
| Exp. permanent damage cases | 13.9 | 3.1 |
| Exp. temporary damage cases | 213.5 | 48.5 |

Since no better information about accidents arising in the construction and operation of these plants is available we evaluate them at the same rates given above. We find it clearer at this stage to compute separately the damage for each plant, leaving calculations on the net impact to a following section.

Hence, the figures of expected accidental damages resulting are those reported below.

Obviously, if these impacts have to be expressed in terms of expected damages per kWh recovered, not only should the analysis account for the yearly electricity production, but also for the plant lifetime production which is taken to be 10 years. If no extraordinary interruption for breakdown is considered, and if average waste calorific power is supposed not to change

during 10 years, the net total electricity output will be 2477 GWh. Total energy recovered during the same time horizon from the alternative landfill disposal considered would lead to a net production of about 300 GWh.

It has to be stressed that these figures do not include maintenance interruptions for plant breakdowns, which are quite likely to happen in such a long time period - although they are not easily predictable. Therefore, the numbers given are likely to underestimate the number of accidents per tonne of waste disposed.

**Figure 3.4** Public health accidents impact for incinerator and landfill construction

| **Impact** | **Incinerator** | **Landfill** |
|---|---|---|
| | (ECU/t waste) | (ECU/t waste) |
| Death cases | 0.71 | 0.05 |
| Permanent damage cases | 0.11 | 0.01 |
| Temporary damage cases | 6E-03 | 6E-04 |
| TOTAL | 0.83 | 0.08 |

### 3.7.2 Incinerator worker's health

Even if there are no *specific* professional illnesses occurring to incinerator workers, they can nevertheless be frequently exposed to dangerous substances, either by contact or by inhalation. However, it is very difficult to predict the kind of health damages that can occur, given the variability of waste composition.

Health damages can occur in cases of occasional leakages of flue gases from the furnaces, and they can be of every kind depending on the nature of the escaped gases, the concentration levels involved, and the period of exposure. Dangerous chemical substances can also occasionally contaminate workers by dermal contact. It happened in the past, e.g., that asbestos wastes were illegally disposed of through incinerators and landfills with municipal wastes, causing cancer cases to some workers. This represents however a quite unlikely and unpredictable case which cannot be evaluated here.

There is of course agreement on the fact that several preventive and monitoring actions should be taken in order to protect incinerator worker's health; however, due to the highly variable nature of wastes, and to the unpredictable nature of the chemical reactions they might occur during combustion or storage, it turns out to be difficult to predict their specific health effects; thus no attempt is made to define an average representative health damage occurring to this particular profession.

The same conclusions may also apply to the alternative (landfill) disposal solution. Differences between a MSW sanitary landfill and a plant for incineration residuals are probably small, as in both cases workers would be exposed to heavy metals, though in a more volatile form in the latter case. However, in the case of a MSW landfill workers are exposed to emissions of biogas at the highest concentration identified in this report for public health impacts. In particular, they would be exposed to risks of asphyxia due to accumulation of carbon dioxide in the lowest points of the landfill; modern working teams are equipped against such risk which is not quantified in the technical literature surveyed.

## 3.8 Municipal solid waste disposal and global warming

The relationship between solid waste disposal and global warming is a complex one and can be looked at in two main different ways.

The first one is a *life cycle* approach where the researcher analyses the global warming impact of the whole life cycle of a product (from cradle to grave, i.e. from production to waste disposal). From this perspective, the carbon balance of a product may be positive, neutral or negative. For instance, the whole cycle of producing , using and finally disposing of a product may well represent an overall net subtraction of carbon from the atmosphere, and the product be regarded as a form of carbon sink.

The approach followed in this paper is quite different. We do not consider the rest of a cycle of products which are disposed of as wastes: whatever the impact of their production and use on global warming, they must be eventually disposed of as wastes. The only relevant point here is that such waste disposal may occur in landfill sites or through incineration, the latter being regarded as a way of producing energy. Hence, the impact on global warming of incineration arises from the *difference* between the GW impact of waste incineration and that of landfilling.

Incineration and landfilling of MSW are both responsible for large but different emissions of greenhouse gases (GhG), mainly carbon dioxide and methane, though with very different emission rates and time patterns. While $CO_2$ emissions from incineration resemble those arising from fossil fuel cycles but for emission rates, emissions of each MSW unit disposed of in landfill sites are distributed across time as biogas production tends to rapidly increase, peak and later gradually decline over a span of 10-20 years. As a consequence of which explicit time-related analysis of emissions and their impacts is required even more than for fossil fuel cycles. In other words, the damage cost of a unit of GhG emissions from a landfill site must be further discounted according to the emission time path of the landfill.

Our treatment of global warming follows the approach used for the ExternE National Implementation Study. Emissions of GhG are multiplied by unit values to get damage estimates.

The next step is discounting of emissions damages from a landfill site in order to account for the emission delay with respect to those of the incinerator. Summing over the delay time yields total damage that can be attributed to GhG emission from landfilling one tonne of MSW. At this stage, the damage may be compared with that arising from incineration of the "same" tonne, obtained by multiplying incinerator GhG emissions by unit values.

Finally, estimation of external costs of waste incineration, regarded as a fuel cycle, is carried out by algebraic summation of impacts from MSW incineration *and* landfilling for the evaluation of MSW incineration as an energy production cycle, where incineration largely displaces GW effects of the landfill.

### 3.8.1 Emissions of greenhouse gases

GhG emissions of each stage of this fuel cycle arising from disposal of one tonne of MSW have been discussed in sections 2.1 and 2.4 and are summarised for convenience in Table 3.11. Transport externalities are calculated by assuming that landfilling involves haulage by trucks for 40 km more than for incineration, as the landfill site is 20 km farther away from the waste generation area. It involves $CO_2$ emissions of 1.52 kg/t·km.

It is assumed that landfill gas consists of constant shares of $CO_2$ (45%) and $CH_4$ (55%), of which 60% is recovered and burned to raise electricity while 40% is dispersed. Other GhG that may be emitted by landfills (e.g. halocarbons) are not considered due to lack of reliable data and the impossibility of satisfactory analysis of their alternative incineration treatment. Total generation of landfill gas amounts to 160 $m^3$/tonne over a 15 year period, distributed across time as in Fig. 2.1.

The total rate of GhG emissions could also be evaluated as 1.398 kg of $CO_2$ and 0.117 kg of $CH_4$ per cubic meter of landfill gas generated. This includes emissions from both dispersion and recovery.

**Table 3.11** Greenhouse gas emissions per tonne of MSW

| GhG | $CO_2$ (kg/tonne) | $CH_4$ (kg/tonne) |
|---|---|---|
| Incineration | 1100.0 | - |
| Landfill gas dispersed | 95.0 | 18.9 |
| Landfill gas recovered | 63.4 | - |
| Landfill gas recovery exhausts | 65.3 | - |

Damage valuation per tonne of MSW is given in Table 3.12, as computed according to the monetary values defined in the ExternE-Core programme, using the "illustrative restricted range" of monetary values (European Commission, 1998). As explained above, delayed

damages due to emissions of GhGs from landfill gas dispersion and recovery are discounted at 1%, which is consistent with assumptions on which the valuation of GW damage is based.

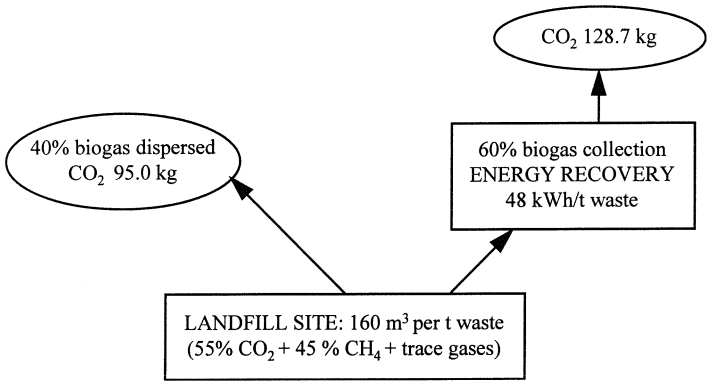



**Figure 3.5** Landfill gas flow scheme

Recall now that incineration of one tonne of MSW yields 888 kWh while landfill gas recovery from the same tonne only 48. Hence incineration as a fuel cycle generates 840 kWh/t(MSW). The *net* damage from one kWh generated by incineration (instead of landfilling, with biogas recovery) is therefore between 10.6 mECU 1995 (lower bound) and 37.7 mECU 1995 (upper bound).

**Table 3.12** Damages per tonne of MSW

| | $CO_2$ (lower) | $CO_2$ (upper) | $CH_4$ (lower) | $CH_4$ (upper) |
|---|---|---|---|---|
| Unit value (ECU 1995/t) | 18 | 46 | 400 | 520 |
| Incineration (ECU/twaste) | 19.80 | 50.60 | - | - |
| Incin. transport (ECU/twaste) | 0.01 | 0.03 | - | - |
| Landfill gas (ECU/twaste) | 3.78 | 9.66 | 7.08 | 9.20 |
| Landfill transport | 0.04 | 0.09 | - | - |

## 3.9 The *net* damage of waste incineration as a fuel cycle

The damages of both waste disposal plants analysed, i.e. the incinerator and the landfill with biogas recovery, are reported separately in Table 3.13 as if they were "stand-alone" technologies, giving the damage in terms of ECU per tonne of waste disposed. Landfill damage was calculated as explained in sec. 3.6, and by Ecosense 2.0 for the impacts of atmospheric pollutants of the gas recovery process.

It does not seem particularly sensible to analyse the landfill as a technology *for energy production* per se, without taking into account what is its first traditional aim – i.e. that of disposing of wastes. Such a particular viewpoint would suggest in fact that, for instance, it is not worth transporting waste to the landfill and treating them *in order to* produce energy, as the external damage of this stage alone turns out to be at least an order of magnitude higher than that of any other power generation fuel cycle. The conclusion can obviously change if we admit that the landfill already exists and operates for other purposes (i.e. waste disposal): electricity production is in this case just an *additional* output of this process.

The final step that has to be completed to interpret our results according to the fuel cycle definition given in the first paragraph, is that of comparing the two technologies – incinerator and landfill – firstly *as waste disposal processes*, giving data in terms of economic value per unit of waste disposed, and computing only in a second stage the *net* external impact of the incinerator for what concerns both waste disposal and energy generation. This comparison is made in Table 3.13, where all categories of damage estimated are couched in terms of external cost per tonne of waste.

Let us leave aside - again, just for a while - the greenhouse gas effects, affected by much greater uncertainty than other damage classes. The table suggests that even including the evaluation of the *differential* transport costs (which is 3.41 ECU/t in favour of the incinerator) and that of the amenity losses for the landfill, waste treatment through incineration causes still a higher external damage: 26.8 ECU/t versus 16.5 for the latter technology.

The conclusion does not change even if we include the global warming effect (taking the lower and upper values of the illustrative restricted range given for $CO_2$ and $CH_4$). The comparison gives 77.4 ECU/t versus 35.4 for the landfill in the upper case and 46.6 versus 27.1 in the lower case.

Looking at the results obtained, the incinerator thus seems to be the most polluting technology. It has to be emphasised, however, that this conclusion has to be taken with much caution. In fact, first of all the evaluations presented have a certain degree of uncertainty. Moreover, the conclusion might change significantly by taking different viewpoints - e.g., if we want to consider also the displaced effects resulting from another, different fuel cycle that produces the same amount of net energy recovered by the incinerator.

**Table 3.13** Damages and avoided costs (*per tonne*) of the waste incineration fuel cycle

| | MSW Incinerator | Landfill |
|---|---|---|
| | **ECU/t** | **ECU/t** |
| **POWER GENERATION** | | |
| Public health (mortality and morbidity), *of which:* | 23.69 | 0.26 |
| *TSP* | *1.01* | *0* |
| $SO_2$ | *3.39* | *0.05* |
| $NO_x$ | *17.65* | *0.19* |
| $NO_x$ *(via ozone)* | *1.62* | *0.02* |
| *Cd-pCDD/F* | 0.02 | nr |
| *Benzene* | *nr* | 7.3E-05 |
| Crops | 0.49 | 0.005 |
| Ecosystems | nq | nq |
| Materials | 0.22 | 0.012 |
| *Monuments* | *nq* | *nq* |
| Noise | nq | nq |
| Amenity losses | ng | 10.89 |
| Global warming *(Illustrative restricted range)* | | |
| $CO_2$ | | |
| low | 19.8 | 3.8 |
| mid | 50.6 | 9.6 |
| $CH_4$ | | |
| low | nr | 7.1 |
| mid | nr | 9.2 |
| **OTHER FUEL CYCLE STAGES** | | |
| Public health | 1.46 | 4.60 |
| *TSP* | *1.44* | *4.55* |
| *SOx* | 0.003 | 0.011 |
| Road accidents | 0.14 | 0.68 |
| Occupational health | 0.84 | 0.08 |
| Ecological effects | ng | ng |
| Road damages | nq | nq |
| Global warming | | |
| low | 0.01 | 0.04 |
| mid | 0.03 | 0.09 |

*Yoll= mortality impacts based on 'years of life lost' approach, VSL= impacts evaluated based on 'value of statistical life' approach.
ng: negligible; nq: not quantified; iq: only impact quantified; nr : not relevant

Considering the first comparison mentioned above, the disposal of one tonne of waste by mass burning causes a damage of 26.8 ECU *less* the displaced effect of 16.5 ECU that would have been caused by the alternative disposal: therefore, a *net damage* of 10.3 ECU/t. According to the technology specification of the incinerator analysed (see Appendix 1) this turns out to be the external cost faced to generate 888 kWh. However, we have to take also into account another displaced effect, which is the amount of energy that would have been generated by the utilisation of the biogas recovered from one tonne of waste disposed in the landfill: 48 kWh.

The *net* energy production per tonne of waste disposed of becomes therefore 840 kWh. With this number, it is finally possible to compute the net external cost *per kWh generated* by the waste incineration fuel cycle, as defined in section 1.1.1: it amounts to 12.3 mECU/kWh. This result is almost three times lower than that obtained considering waste incineration as a technology per se, aimed at energy generation. It is also interesting to notice that this result is of the same order of magnitude as the external costs of power generation from a natural gas plant.

# 4. CONCLUSIONS

The present study evaluates the environmental externalities due to energy recovery from waste incineration according to the ExternE methodology. Great care was put at the outset of this work in the definition of what is the "fuel cycle" that has to be evaluated. In fact, this technology has some peculiarities with respect to the "traditional" fuel cycles previously analysed (see EC DG XII, "ExternE", 1995b, 1995c, 1995d) which deserve special attention. In particular, emphasis was put on the main distinguishing feature of the "fuel" analysed: waste is not an economic good produced with the specific aim of burning it, but rather an economic "bad", generated from any production and consumption activity, that must be disposed of in some way.

This fundamental feature led us to delineate a fairly complex fuel cycle definition as a basis for the following steps of the analysis.

First of all, waste incineration has to be assessed against a "benchmark", given by a suitable alternative waste treatment technology. This ultimately means that in addition to the usual external impact of the process studied, the *avoided* impact that would have arisen from the alternative disposal process has to be evaluated and taken into account (obviously with a *minus* sign). The benchmark technology chosen in this case was a landfill plant equipped with biogas recovery and electricity generation facilities.

This also means that some of the fuel cycle stages usually considered for other technologies have in this case to be treated differently. For instance, it would not be sensible in our case to assess the externalities arising from a "fuel production" stage. Moreover, we hold that even the "waste transport" stage cannot properly be considered as part of the waste incineration fuel cycle - in particular when this is seen as an energy recovery technology. In fact, it cannot be claimed that wastes are transported to the incinerator *in order to* produce electricity from them. It is more correct to state that wastes are transported *away from* the premises where they have been originated, first of all because of hygienic needs. It would therefore be misleading to input waste transport externalities to the electricity production, couching the corresponding results in terms of mECU/kWh. It might be sensible, on the other hand, to calculate the *differential* impact arising from transport to the two alternative plants evaluated. There is, in fact, a systematic difference in the distance of incinerators and landfills from the towns that originate wastes. The former, which need less space to be built, are usually located much closer to the town centre then the latter.

The whole report has the final aim of calculating the *net* impact of energy recovery from waste incineration minus the avoided impact of landfill disposal of the same wastes. In order to

obtain this result, both technologies were first evaluated separately, and finally compared. Both separate and net impacts are given in the report.

The peculiarities of the fuel cycle analysed with respect to most of the other technologies previously evaluated do not attain to the fuel cycle definition only. There are also, in fact, some relevant impact pathways that require some extensions of the basic ExternE methodology. First of all, the evaluation of the impact arising from carcinogenic persistent micropollutants brought us to examine and describe (in some cases only qualitatively) all the possible direct and indirect pathways (e.g. inhalation, dermal contact, ingestion...) through which they can eventually affect human health. This has represented quite a new issue with respect to previous ExternE reports.

Secondly, for the sake of completeness, the analysis of welfare losses due to the visual and olfactory public perception of the landfill site was included. This evaluation was achieved through an hedonic pricing study made for the site of Cerro Maggiore.

There are still some impacts which could not be quantified due to lack of data, and others whose evaluation would require considerable improvements. The damage that may arise in case of landfill leaching was not assessed, but it is expected to be low for modern landfill plants which, unlike the old ones, are almost perfectly isolated from surrounding land and subject to leachate drainage and treatment, the cost of which is internalised. More serious dangers may derive from abuse of sites, e.g. disposal of hazardous materials that may cause liner ruptures and leakages from plants designed for MSW. It is however dubious to whom such damages should be attributed.

Damages to materials were quantified based on the Ecosense database, which is referred to North European buildings, and is therefore poorly representative of the Italian reality. Material damage has proved to be an order of magnitude lower than human health damage, so that even a biased estimation of this issue is not likely to change the final results significantly.
Finally, there are other atmospheric micropollutants emitted either by the incinerator (e.g. arsenic, chromium) or by the landfill that are likely to contribute to human health damage, but were not considered in this report. The effect of micropollutants like dioxins, cadmium, mercury and lead has also proven, however, to be generally much lower than that of macropollutants. Even the inclusion of more micropollutants in the analysis is not likely to change significantly our results.

The values found in this study seem to suggest that environmental externalities from MSW incineration exceed those due to MSW landfilling. There are however several reasons why this should not be taken as a general conclusion, nor should it be easily used for policy and planning purposes.

Firstly, results (though not the methodology) are specific for the two sites analysed, and cannot be easily transferred to other realities. Secondly, it is worth mentioning - although this is quite an obvious observation - that our study only treats environmental externalities. Other, very important economic variables that determine the choices among different disposal

options (and might change the relative appeal of incinerator and landfill sites that is shown in our partial analysis) were not considered in the report, being beyond the scope of the research.

Finally, the policy maker might also look at the results presented here from a different perspective, which is slightly more favourable to the incinerator technology compared to our conclusions. In fact, if we consider that the energy which is not recovered by these plants would have to be produced by other fuel cycles, we end up with another, "displacement effect" that decreases the net external impact of both technologies (and that of the incinerator in particular). This displacement effect can obviously vary considerably according to the fuel cycle which is displaced. This view, which is adopted in the UKDOE (1993) study, is however not consistent with the approach taken in the ExternE studies, which is shared by the present report. Our aim (reflecting the fact that the ExternE project is a research study concerned mainly with development of a suitable methodology for external costs evaluation) was in fact to evaluate the fuel cycle per se. This leaves to the policy maker any consideration on the interactions (included possible displacement effects) between different fuel cycles generation.

# APPENDIX 1. - DEFINITION OF THE WASTE INCINERATION TECHNOLOGY

| Stage | Parameter | Quantity | Source of data, comments |
|---|---|---|---|
| **1.Waste transport** | Origin | Provisional store in Milano, v. Olgettina | |
| | Destination | Incinerator of Milano-Silla | |
| | Distance (return journey) | 10 km | |
| | Alternative destination for MSW disposal | Landfill of Cerro M. | |
| | Distance (return journey) | 50 km | |
| | Mode of transport | Road (100%) | |
| | Vehicle | Heavy duty diesel vehicle | |
| | Capacity | 15 t. | |
| | MSW weight | 279000 t/yr | |
| | trips per year | 18600 | |
| | Air emissions | | |
| | *TSP* | 0.15 t/yr | |
| | $NO_x$ | 1.38 t/yr | |
| | $SO_x$ | 0.03 t/yr | |
| | *CO* | 1.36 t/yr | |
| | *VOC* | 0.14 t/yr | |
| | $CO_2$ | 110.2 t/yr | |
| **2. Power generation** | Waste composition (in weight) | | |
| | Paper and cardboard | 31% | |
| | Organic matter | 29% | |
| | Plastic | 17% | |
| | glass and inert matter | 2% | |
| | wood | 1.4% | |
| | metals | 0.4% | |
| | clothes | 5.2% | |
| | napkins | 3.5% | |
| | other | 10.5% | |
| | calorific power | 15000 kJ/kg | |
| | combustion technology | Grate-type furnace | |
| | efficiency (all electric) | 21% | |
| | net power | 33.3 MW | |
| | load hours | 7440 | |
| | lifetime | 10 yr | |
| | pollution abatement technology | Selective non-catalytic reduction | |
| | stack height | 100 m. | |
| | stack diameter | 3 m. | |
| | Flue gas volume | 250000 $Nm^3/h$ | |

| Stage | Parameter | Quantity | Source of data, comments |
|---|---|---|---|
| | air emissions | | |
| | *TSP* | 5.58 t/yr | |
| | $NO_x$ | 372 t/yr | |
| | $SO_x$ | 93 t/yr | |
| | $CO_2$ | 306900 t/yr | |
| | *Cd* | 18.6 kg/yr | |
| | *PCDD/PCDF* | 9.3E-05 kg/yr | |
| | Solid residuals (ashes) | 83700 t/yr | |
| **3. Ashes transport** | Destination | Landfill of Cerro M. | |
| | Distance | 20 km | |
| | Mode of transport | Road (100%) | |
| | Vehicle | Heavy duty diesel vehicle | |
| | Capacity | 15 t. | |
| | MSW weight | 83700 t/yr | |
| | trips per year | 5580 | |
| | Air emissions | | |
| | *TSP* | 0.09 t/yr | |
| | $NO_x$ | 0.83 t/yr | |
| | $SO_x$ | 0.02 t/yr | |
| | *CO* | 0.81 t/yr | |
| | *VOC* | 0.08 t/yr | |
| | $CO_2$ | 66.15 t/yr | |
| **4. Ashes disposal / Benchmark disposal technology** | Location | Cerro Maggiore | |
| | Type of facility | Landfill with biogas recovery and energy production | |
| | Capacity (MSW/day) | 2000 t/day | |
| | biogas generation | 160 $m^3/t$ | |
| | biogas recovery | 96 $m^3/t$ | |
| | biogas calorific power | 3700 $kcal/m^3$ | |
| | combustion technology | Otto cycle engine | |
| | net efficiency | 11.7% | |
| | net power | 4.0 MW | |
| | load hours | 7500 | |
| | stack height | 20 m. | |
| | stack diameter | 0.6 m. | |
| | air emissions | | |
| | *TSP* | 0 | |
| | $NO_x$ | 8.2 t/yr | |
| | $SO_x$ | 1.8 t/yr | |
| | $CO_2$ *(average emission over 15 yr)* | 9237 t/yr | |
| | $CH_4$ *(average emission over 15 yr)* | 780 t/yr | |

# APPENDIX 2. - QUANTIFICATION OF BURDENS FOR THE WASTE INCINERATION PROCESS

| Stage | Burden | Quantity | Source of data | Impact assessed? |
|---|---|---|---|---|
| **1. Waste transport** | | | | |
| | Occupational health | | | |
| | accidents - fatal | nq | | x - no data |
| | accidents - major injury | nq | | x - no data |
| | accidents - minor injury | nq | | x - no data |
| | noise levels | nq | | x - no data |
| | Public health | | - | |
| | accidents - fatal | 0.012 cases/yr | ISTAT (1995) | ✓ |
| | accidents - major injury | nq | | |
| | accidents - minor injury | nq | | |
| | Air emissions (transport) | | Corinair (1991) | |
| | $CO_2$ | 110.2 t/yr | | ✓ |
| | $CH_4$ | nq | | x - no data |
| | CO | 1.36 t/yr | | ✓ |
| | $SO_2$ | 0.03 t/yr | | ✓ |
| | $NO_x$ | 1.38 t/yr | | ✓ |
| | $PM_{2.5}$ | 0.15 t/yr | | ✓ |
| | VOC | 0.14 t/yr | | ✓ |
| | Traffic congestion | nq | | x - no data |
| | Noise | nq | | x - no data |
| | Odours | nq | | x - no data |
| **2. Power generation** | Occupational health | | | |
| | accidents - fatal | nq | | x - no data |
| | accidents - major injury | nq | | x - no data |
| | accidents - minor injury | nq | | x - no data |
| | Air emissions | | | ✓ |
| | $CO_2$ | 306900 t/yr | | ✓ |
| | $SO_2$ | 93 t/yr | | ✓ |
| | $NO_x$ | 372 t/yr | | ✓ |
| | $PM_{10}$ | 5.58 t/yr | | ✓ |
| | HCl + HF | 18+1,86 t/yr | | x - no data |
| | Heavy Metals | | | x - no data |
| | *of which Cd* | 18.6 kg/yr | | ✓ |
| | *Hg + Pb* | 0.03+0.01 t/yr | | ✓ |
| | PCDD/PCDF | 9.3E-05 kg/yr | | ✓ |
| | Amenity losses | - | | ✓ |
| **3. Transmission** | No additional burdens | | | |
| **4. Transport of solid residuals (ashes and slag)** | | | | |
| | Occupational health | | | |
| | accidents - fatal | nq | | x - no data |
| | accidents - major injury | nq | | x - no data |

| Stage | Burden | Quantity | Source of data | Impact assessed? |
|---|---|---|---|---|
| | accidents - minor injury | nq | | x - no data |
| | Public health | | | |
| | accidents - fatal | 0.007 cases/yr | ISTAT (1995) | ✓ |
| | accidents - major injury | nq | | x - no data |
| | accidents - minor injury | nq | | x - no data |
| | Air emissions (vehicle) | | Corinair (1991) | |
| | $CO_2$ | 66.15 t/yr | | ✓ |
| | $CH_4$ | nq | | x - no data |
| | CO | 0.81 t/yr | | ✓ |
| | $SO_2$ | 0.02 t/yr | | ✓ |
| | $NO_x$ | 0.83 t/yr | | ✓ |
| | $PM_{2.5}$ | 0.09 t/yr | | ✓ |
| | VOC | 0.08 t/yr | | ✓ |
| | Air emissions (fugitive dust) | | | ✓ |
| | $PM_{10}$ | nq | | x - no data |
| | *of which heavy metals* | nq | | x - no data |
| | Traffic congestion | nq | | x - no data |
| | Noise | nq | | x - no data |
| **5. Ashes disposal –** | Occupational health | | | |
| | accidents - fatal | nq | | x - no data |
| | accidents - major injury | nq | | x - no data |
| | accidents - minor injury | nq | | x - no data |
| | Air emissions from activities on site | | | |
| | $CO_2$ | nq | | x - no data |
| | $CH_4$ | nq | | x - no data |
| | CO | nq | | x - no data |
| | $SO_2$ | nq | | x - no data |
| | $NO_x$ | nq | | x - no data |
| | $PM_{10}$ - combustion | nq | | x - no data |
| | $PM_{10}$ - fugitive dust | nq | | x - no data |
| | VOC | nq | | x - no data |
| | Leachate | nq | | x - no data |
| | Visual impact (landfill) | | Ascari et al. (1996) | ✓ |
| | Noise | nq | | x - no data |
| **6. Construction** | Occupational health | | | |
| | accidents - fatal | 0.2 cases/TWh | IEFE (1995) | ✓ |
| | accidents - major injury | 5.6 cases/TWh | | ✓ |
| | accidents - minor injury | 86 cases/TWh | | ✓ |
| | Air emissions from materials production | | | |
| | $CO_2$ | nq | | x - no data |
| | $SO_2$ | nq | | x - no data |
| | $NO_x$ | nq | | x - no data |
| | $PM_{10}$ - combustion | nq | | x - no data |
| | Air emissions from materials transport | | | |

| Stage | Burden | Quantity | Source of data | Impact assessed? |
|---|---|---|---|---|
| | $CO_2$ | nq | | x - no data |
| | $SO_2$ | nq | | x - no data |
| | $NO_x$ | nq | | x - no data |
| | $PM_{10}$ - combustion | nq | | x - no data |
| | Air emissions from activities on site | | | |
| | $CO_2$ | nq | | x - no data |
| | $SO_2$ | nq | | x - no data |
| | $NO_x$ | nq | | x - no data |
| | $PM_{10}$ - combustion | nq | | x - no data |
| | $PM_{10}$ - fugitive dust | nq | | x - no data |
| | Noise | nq | | x - no data |
| | Road use | nq | | x - no data |
| **11. Demolition** | Occupational health | | | |
| | accidents - fatal | nq | | x - no data |
| | accidents - major injury | nq | | x - no data |
| | accidents - minor injury | nq | | x - no data |
| | Air emissions from materials production | | | |
| | $CO_2$ | nq | | x - no data |
| | $SO_2$ | nq | | x - no data |
| | $NO_x$ | nq | | x - no data |
| | $PM_{10}$ - combustion | nq | | x - no data |
| | $PM_{10}$ - fugitive dust | nq | | x - no data |
| | Noise | nq | | x - no data |
| | Road use | nq | | x - no data |

# APPENDIX 3. - QUANTIFICATION OF IMPACTS AND DAMAGES FOR THE WASTE INCINERATION PROCESS

**Impacts and damages from the waste incineration fuel cycle (Milano – Silla) before subtraction of landfill avoided damage**

| Impact | *Impact - units/TWh* | Impacts - number | Damages mECU/kWh | *ECU/$t_{poll}$* | $\sigma_g$, range |
|---|---|---|---|---|---|
| **Air pollution** | | | | | |
| **Power generation** | | | | | |
| **Primary $PM_{10}$** | | | 1.1 | 50506 | A |
| Acute mortality | YOLL | 0.08 | 0.01 | | |
| | deaths | 0.11 | 0.33 | | |
| Chronic mortality | YOLL | 11.8 | 0.93 | | |
| | deaths | 1.18 | | | |
| Acute and chronic morbidity | cases | | 0.16 | | |
| Materials - cleaning costs | | | | | |
| **$NO_x$ - all from nitrate aerosol** | | | 19.9 | 13244 | A |
| Acute mortality | YOLL | 1.51 | 0.23 | | |
| | deaths | 2.01 | 6.25 | | |
| Chronic mortality | YOLL | 208.8 | 17.6 | | |
| | deaths | 20.8 | | | |
| Acute and chronic morbidity | cases | | 2.28 | | |
| Materials damage - utilitarian | | | | | |
| Materials damage - cultural | | | | | |
| Critical loads exceedance | | | | | |
| Effects on crop yield | | | | | |
| **$SO_2$ - as sulphate** | | | 3.8 | 10173 | A |
| Acute mortality | YOLL | 0.30 | 0.05 | | |
| | deaths | 0.39 | 1.22 | | |
| Chronic mortality | YOLL | 40.23 | 3.39 | | |
| | deaths | 4.23 | | | |
| Acute and chronic morbidity | cases | | 0.43 | | |
| Materials damage – utilitarian ($SO_2$) | | | 0.25 | | |
| Materials damage – cultural ($SO_2$) | | | n.q. | | |
| Critical loads exceedance | | | n.q. | | |
| Effects on crop yield ($SO_2$) | | | 0.002 | | |
| **$O_3$** | | | | 1494 | B |
| Acute mortality | | | 0.66 | 412 | |
| Acute morbidity | | | 1.17 | 732 | |
| Effects on crop yield | | | 0.56 | 350 | |
| **Amenity losses** | - | - | - | ng | B |
| **Greenhouse gas emissions** | | | | | C |
| $CO_2$ | - | - | | | |
| low | | | 8.4 | 6.8 | |
| mid | | | 57.0 | 46 | |
| high | | | 107.8 | 87 | |
| $N_2O$ | - | - | nq | | |
| $CH_4$ | - | - | nq | | |

| Impact | *Impact - units/TWh* | Impacts - number | Damages mECU/kWh | *ECU/$t_{poll}$* | $\sigma_g$, range |
|---|---|---|---|---|---|
| ***Waste transport*** | | | | | |
| **Primary $PM_{10}$** | | | 1.03 | 3974280 | A |
| Chronic mortality | YOLL | 10.59 | 0.89 | | |
| Acute and chronic morbidity | cases | 701.1 | 0.12 | | |
| **NOx (as nitrates)** | | nq | nq | nq | |
| **SOx (as sulphates)** | | nq | nq | nq | |
| **$O_3$** | | | 9.1E-03 | | B |
| Acute mortality | | | 2.6E-03 | | |
| Acute morbidity | | | 4.6E-03 | | |
| Effects on crop yield | | | 1.9E-03 | | |
| **Greenhouse gas emissions** | | | | | C |
| $CO_2$ | | | | | |
| low | - | - | 3.1E-03 | 6.8 | |
| mid | | | 2.0E-02 | 46 | |
| high | | | 3.8E-02 | 87 | |
| $N_2O$ | - | - | nq | nq | |
| $CH_4$ | - | - | nq | nq | |
| **Road accidents** | | | | | |
| deaths | VSL | 0.05 | 0.15 | - | A |
| ***Ashes and slag transport*** | | | | | |
| **Primary $PM_{10}$** | | | 0.62 | 3974280 | |
| Chronic mortality | YOLL | 6.35 | 0.54 | | |
| Acute and chronic morbidity | cases | 420 | 0.07 | | |
| **NOx (as nitrates)** | | nq | nq | nq | |
| **SOx (as sulphates)** | | nq | nq | nq | |
| **$O_3$** | | | | | B |
| Acute mortality | | | 1.5E-03 | | |
| Acute morbidity | | | 2.7E-03 | | |
| Effects on crop yield | | | 1.1E-03 | | |
| **Greenhouse gas emissions** | | | | | C |
| $CO_2$ | | | | | |
| low | - | - | 1.8E-03 | 6.8 | |
| (upper value of restricted range) | | | 1.2E-02 | 46 | |
| high | | | 2.3E-02 | 87 | |
| $N_2O$ | - | - | nq | nq | |
| $CH_4$ | - | - | nq | nq | |
| **Road accidents** | | | | | |
| deaths | cases | 0.03 | 0.09 | - | A |
| **Construction** | | | | | |
| **Occupational health effects** | | | | | B |
| **1. Construction** | | | | | |
| Fatalities | VSC | 0.26 | 0.8 | - | |
| Major injuries | cases | 5.6 | 0.13 | - | |
| Minor injuries | cases | 86.2 | 0.007 | - | |

# APPENDIX 4. EXPOSURE-RESPONSE FUNCTIONS USED FOR QUANTIFICATION OF HEALTH DAMAGES

Quantification of health impacts for the *ExternE Core Project*. The exposure response slope, $f_{er}$ , has units of cases / (yr-*receptor*-μg/m$^3$). The cost is in 1995 ECU per case.

| Receptor | Impact Category | Reference | Pollutant | $f_{er}$ | Cost | Uncertainty rating[ii] |
|---|---|---|---|---|---|---|
| **ASTHMATICS** (1.5% of population) | | | | | | |
| *adults* | Bronchodilator usage | Dusseldorp et al., 95 | Nitrates | 0.163 | 37 | B |
| | | | $PM_{10}$ | 0.163 | | B |
| | | | Sulfates | 0.272 | | B |
| | | | $PM_{2.5}$[iii] | 0.272 | | B |
| *adults* | Cough | Dusseldorp et al., 95 | Nitrates | 0.168 | 7 | A |
| | | | $PM_{10}$ | 0.168 | | A |
| | | | Sulfates | 0.280 | | A |
| | | | $PM_{2.5}$ | 0.280 | | A |
| *adults* | Lower respiratory symptoms (wheeze) | Dusseldorp et al., 95 | Nitrates | 0.061 | 7.5 | A |
| | | | $PM_{10}$ | 0.061 | | A |
| | | | Sulfates | 0.101 | | A |
| | | | $PM_{2.5}$ | 0.101 | | A |
| *children* | Bronchodilator usage | Roemer et al., 93 | Nitrates | 0.078 | 37 | B |
| | | | $PM_{10}$ | 0.078 | | B |
| | | | Sulfates | 0.129 | | B |
| | | | $PM_{2.5}$ | 0.129 | | B |
| *children* | Cough | Pope/Dockery, 92 | Nitrates | 0.133 | 7 | A |
| | | | $PM_{10}$ | 0.133 | | A |
| | | | Sulfates | 0.223 | | A |
| | | | $PM_{2.5}$ | 0.223 | | A |
| *children* | Lower respiratory symptoms (wheeze) | Roemer et al., 93 | Nitrates | 0.103 | 7.5 | A? |
| | | | $PM_{10}$ | 0.103 | | A |
| | | | Sulfates | 0.172 | | A |
| | | | $PM_{2.5}$ | 0.172 | | A |
| **ELDERLY 65+** | | | | | | |
| | Congestive heart | Schwartz/Morris, 95 | Nitrates | 1.85E-5 | 7870 | B |
| | | | $PM_{10}$ | 1.85E-5 | | B |

| | | | | | | |
|---|---|---|---|---|---|---|
| | failure | | Sulfates | 3.09E-5 | | B |
| | | | $PM_{2.5}$ | 3.09E-5 | | B |
| | | | CO | 5.64E-7 | | B |
| **Receptor** | **Impact Category** | **Reference** | **Pollutant** | **$f_{er}$** | **Cost** | **Uncertainty rating** |
| **CHILDREN** | | | | | | |
| | Chronic bronchitis | Dockery et al., 89 | Nitrates | 1.61E-3 | 225 | B |
| | | | $PM_{10}$ | 1.61E-3 | | B |
| | | | Sulfates | 2.69E-3 | | B |
| | | | $PM_{2.5}$ | 2.69E-3 | | B |
| | Chronic cough | Dockery et al., 89 | Nitrates | 2.07E-3 | 225 | B |
| | | | $PM_{10}$ | 2.07E-3 | | B |
| | | | Sulfates | 3.46E-3 | | B |
| | | | $PM_{2.5}$ | 3.46E-3 | | B |

[5]The UKDOE (1993) report claims that disamenity effects are independent of plants size, but this seems not justified and at odds with our empirical results. However some fixed damage effect not related to landfill size may exist.

[6]Odour emissions abatement practices which are more easily implemented after plant closure and the existence of a perceptive threshold further complicates the requested correction.

[7]Uncertainty rating: A = high confidence ( $\sigma_g$ = 2.5 - 4)
B = medium confidence ( $\sigma_g$ = 4 - 6)
C = low confidence ( $\sigma_g$ = 6 - 12)

| **Receptor** | **Impact Category** | **Reference** | **Pollutant** | **$f_{er}$** | **Cost** | **Uncertainty rating** |
|---|---|---|---|---|---|---|
| **ADULTS** | | | | | | |
| *80% of population* | Restricted activity days (RAD) | Ostro, 87 | Nitrates | 0.025 | 75 | B |
| | | | $PM_{10}$ | 0.025 | | B |
| | | | Sulfates | 0.042 | | B |
| | | | $PM_{2.5}$ | 0.042 | | B |
| *80% of population* | Chronic bronchitis | Abbey et al., 95 | Nitrates | 4.9E-5 | 105000 | A |
| | | | $PM_{10}$ | 4.9E-5 | | A |
| | | | Sulfates | 7.8E-5 | | A |
| | | | $PM_{2.5}$ | 7.8E-5 | | A |
| *57% of population (> 30 yrs)* | Mortality (*'chronic' Yoll*) | Pope et al., 95 | Nitrates | 7.2E-4 | 84330 [4] | B |
| | | | $PM_{10}$ | 7.2E-4 | | B |
| | | | Sulfates | 1.2E-3 | | B |
| | | | $PM_{2.5}$ | 1.2E-3 | | B |

| | | | | | | |
|---|---|---|---|---|---|---|
| **ENTIRE POPULATION** | | | | | | |
| | Respiratory hospital admissions (RHA) | Dab et al., 96 | Nitrates<br>$PM_{10}$<br>Sulfates<br>$PM_{2.5}$ | 2.07E-6<br>2.07E-6<br>3.46E-6<br>3.46E-6 | 7870 | A<br>A<br>A<br>A |
| | Cerebrovascular hospital admissions | Wordley et al., 97 | Nitrates<br>$PM_{10}$<br>Sulfates<br>$PM_{2.5}$ | 5.04E-6<br>5.04E-6<br>8.42E-6<br>8.42E-6 | 7870 | B<br>B<br>B<br>B |
| | Acute Mortality (*'acute' Yoll*) | London/Athens, 96 (case studies) | $SO_2$ | 5.34E-6 | 155000[5] | B |
| | Cancers | Pilkington et al., 97a and 97b. | Benzene<br>(fatal)<br>(non-fatal) | 8.00E-8<br>3.43E-8<br>1.29E-3 | 206000<br>0<br>450000 | B<br>B<br>B |

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| | | | | | | |
|---|---|---|---|---|---|---|
| **ENTIRE POPULATION** | | | | | | |
| | Respiratory hospital admissions (RHA) | Dab et al., 96 | Nitrates<br>$PM_{10}$<br>Sulfates<br>$PM_{2.5}$ | 2.07E-6<br>2.07E-6<br>3.46E-6<br>3.46E-6 | 7870 | A<br>A<br>A<br>A |
| | Cerebrovascular hospital admissions | Wordley et al., 97 | Nitrates<br>$PM_{10}$<br>Sulfates<br>$PM_{2.5}$ | 5.04E-6<br>5.04E-6<br>8.42E-6<br>8.42E-6 | 7870 | B<br>B<br>B<br>B |
| | Acute Mortality (*'acute' Yoll*) | London/Athens, 96 (case studies) | $SO_2$ | 5.34E-6 | 155000 [5] | B |
| | Cancers | Pilkington et al., 97a and 97b. | Benzene<br>(fatal)<br>(non-fatal) | 8.00E-8<br>3.43E-8<br>1.29E-3 | 206000<br>0<br>450000 | B<br>B<br>B |

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