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ExternE National Implementation Finland

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Abstract

ExternE National Implementation is a continuation of the ExternE Project, funded in part by the European Commission's Joule III Programme. This study is the result of the ExternE National Implementation Project for Finland. Three fuel cycles were selected for the Finnish study: coal, peat and wood-derived biomass, which together are responsible for about 40% of total electricity generation in Finland and about 75% of the non-nuclear fuel based generation.

The estimated external costs or damages were dominated by the global warming (GW) impacts in the coal and peat fuel cycles, but knowledge of the true GW impacts is still uncertain. From among other impacts that were valued in monetary terms the human health damages due to airborne emissions dominated in all the three fuel cycles. Monetary valuation for ecosystem impacts is not possible using the ExternE methodology at present.

The Meri-Pori power station representing the **coal** fuel cycle is one of the world's cleanest and most efficient coal-fired power plants with a condensing turbine. The coal is imported mainly from Poland. The estimated health damages were about 4 mECU/kWh, crop damages an order of magnitude lower and damages caused to building materials two orders of magnitude lower. The power stations of the **peat** and **biomass** fuel cycles are of CHP type, generating electricity and heat for the district heating systems of two cities. Their fuels are of domestic origin. The estimated health damages allocated to electricity generation were about 5 and 6 mECU/kWh, respectively. The estimates were case-specific and thus an generalisation of the results to the whole electricity generation in Finland is unrealistic. Despite the uncertainties and limitations of the methodology, it is a promising tool in the comparison of *similar kinds* of fuel cycles, new power plants and pollution abatement technologies and different plant locations with each other.

Foreword

There is a growing awareness that decision making related to fuel and technology choice for power generation should take into account all the costs, both internal and external. This is reflected in a large number of EU documents. For instance, the European Commission’s Green Paper "For a European Union Energy Policy" states that the internalisation of external costs is central to energy and environmental policy. Hence, an EU wide common approach to the quantification of these externalities as well as a common understanding of their interpretation for policy and decision making is an important prerequisite for this internalisation.

The first important step to this purpose was made by the EC between 1991 and 1995 with the development of a methodology to evaluate the externalities associated with power generation. The ExternE Project, launched within the JOULE I RTD Programme, produced a consistent "bottom-up" accounting framework demonstrated it for the most important fuel cycles. It has since then been widely recognised as the most developed methodology to account externalities of power generation.

The next step was to develop an adequate set of external cost data for different fuel cycles, technologies and countries, as well as to build up expertise in all the member states to assist policy and decision makers in the use of these results. Therefore, within JOULE III the ExternE National Implementation Project was organised. During 1996 and 1997 research teams within all member states of the EU (except Luxembourg) and Norway implemented the ExternE accounting framework to a large number of individual fuel cycles for power generation. Parallel to this project, the methodology was further developed and updated and this was integrated within these data. Thus for the first time, a broad set of comparable data on external costs of power generation is now available. These data take account of site, technology and fuel cycle specificity and this set of data provide a representative overview for electricity generation in the EU. In addition, first estimates for the power generation sector as a whole have been developed.

This publication by **VTT Energy** reports in detail the national implementation in **Finland.** Similar reports have been produced for all countries involved and they all follow the same structure, both to clearly indicate consistency between the different country reports as to ease comparison. These national publications are complemented with publications by the EC on methodology and a summary overview of results for all countries.

The results for the different countries show the importance of technology, fuel and site specificity. This confirms that the approach taken by the EC is the correct one and that the big effort to develop a ‘bottom-up’ methodology and generate a broad set of data is well justified. Energy and environmental policy will only be really efficient and successful if it takes this specificity into account.

The project integrates existing scientific information from different areas and disciplines in a coherent framework. This work could only be successful thanks to the support of the JOULE

Energy RTD program of the EC, to the collaboration of the different research teams in all the countries and to the inputs from a large number of different research programmes both at the EU and national level, and also to the institutes and national public authorities that co-financed this exercise for their contributions to this important work.

The EC JOULE III programme continues to support a further development of the ExternE Project. For the next two years, it will focus on the application of energy use in transport and in this context the ExternE methodology will continue to integrate new scientific developments in the different areas.

The final step towards internalisation relates to the use of the data in policy and decision making. Over the years we have noted a growing interest from research, policy and industry for our results at national and international level. The EC services have now started to feed the ExternE numbers into the policy preparation process for energy, environmental and research policies and the EC-strategy to combat acidification and climate change have profited from this research. This illustrates that notwithstanding all the caveats, these numbers are useful and credible if presented and used in the right context.

It is this new series of data for all countries — supported by the expertise in the country to further develop and exploit these data — that may result in multiple uses of these data for policy and decision making at the level of a country or a region and for both public authorities and the industries. In the end, the real benefit of this research agenda is to be measured by its contribution towards a more sustainable energy use.

Foreword of the Finnish team

VTT Energy was the main contractor of the ExternE National Implementation (NI) in Finland. The work was financed by the European Commission in the framework of the Joule III Programme, by the SIHTI II programme of Technology Development Centre Finland (TEKES) and Technical Research Centre of Finland (VTT) itself. Pekka Pirilä was the project leader and Kim Pingoud the practical coordinator of the project. Kim Pingoud was also responsible for most of the calculations and for the writing of this report. The technical data of the whole coal fuel cycle were supplied by Ekono Energy Ltd. Tomas Otterström was the leader of this subproject, the work being performed by Sari Siitonen at Ekono Energy. Kim Pingoud and Sari Siitonen were responsible for the analysis of the coal fuel cycle, Helena Mälkki for the peat fuel cycle, Margareta Wihersaari for the biomass fuel cycle and Kim Pingoud for the aggregation of the results. Antti Lehtilä wrote the description of the Finnish energy and electricity generation sector and Margareta Wihersaari the country description. Mikko Hongisto from Imatran Voima Oy (and VTT Chemical Technology) presented important critical notes on the ExternE methodology. Matti Johansson from the Finnish Environment Institute assessed the ExternE methodology for ecosystem impacts and compared the results of this NI study with independent Finnish studies.

The following report consists of two kinds of material: 1) the basic theoretical material on the ExternE methodology and a description of the whole ExternE National Implementation, produced by the ExternE Core Project (i.e. the methodology subproject of ExternE), and 2) the description of the Finnish case studies and critical comments on the ExternE methodology, produced by the Finnish team on the basis of experiences of its application.

The aim of the ExternE National Implementation Project was that all the 15 European teams would present their results in a consistent and comparable way and that the reports would have similar structure. Furthermore, it was decided to present the theoretical and methodological material in all the reports uniformly, using the ExternE Core. The report on the national implementation in Finland attempts to meet these requirements. The basic numerical results of the case studies and their aggregation are presented in standard format.

However, within the ExternE National Implementation there appeared to be different conceptions and emphases regarding the methodology, which was understandable in such a large project. Scepticism towards the validity of the methodology, its practical applicability and the generalisations made on the basis of the case studies appeared to be rather strong in the Finnish team. This is also reflected in the structure and content of the Finnish report. It was decided to publish the centrally delivered material unaltered, and the Finnish comments and criticism are presented in a separate chapter and in the conclusion. The numerical results of the Finnish case studies are presented as agreed among the teams in the ExternE NI, but in our opinion extreme caution should be exercised in drawing any quantitative conclusions from the given damage numbers. As a consequence of this choice the report may appear somewhat incoherent internally. The results of the cases and their aggregation to the country level had to be presented in monetary terms due to the original aims of the project. On the other hand, we also wanted to express our own point of view, which partly contradicts our numerical results.

The structure of the report is as follows. First an executive summary of the report is presented. Chapter 1 describes shortly the background and objectives of the whole international project as well as the Finnish National Implementation. Chapter 2 is the presentation of the ExternE methodology written by the ExternE Core Project and is identical in all the 15 national reports. The critical methodological comments by the Finnish team, which are not fully congruent with the standpoints of the preceding chapter, are given in Chapter 3. The numerical results of the Finnish cases and their aggregation are presented in Chapters 4–7. Chapter 8 summarises the main critical conclusions of our work. The material of Appendices I to VII, produced by the Core Project, describes in detail the tools and methodological bases of the ExternE National Implementation Project. More detailed information on the Finnish fuel cycle cases is provided in Appendices VIII, IX and X.

Kim Pingoud

December 1998

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The authors would also like to thank all the members of the 15 national teams in Europe and the experts of the ExternE Core Project, whose guidance was an important factor that allowed the national teams to achieve comparable results.

Dara Connolly and the whole Irish team were in close collaboration with the Finnish team especially in the study of the peat fuel cycle. Pedro Linares, Julio Montes, Rosa Sáez and the others of CIEMAT and Leo de Nocker of VITO successfully coordinated the work in the national implementation projects and brought together all their results into a summary report representing a coherent body of work. Special thanks go to the members of the national teams who hosted the ExternE workshops.

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APPENDICES

Non-technical summary

Introduction

Background and objectives

Externalities are environmental or social impacts of energy production, the damages of which have typically not been reflected in the market price of energy, or considered by energy planners, and consequently have tended to be ignored. Within the European Commission R&D Programme Joule II, the ExternE Project developed and demonstrated a unified methodology for quantification of the externalities of different power generation technologies. It was launched as the EC-US Fuel Cycles Study in 1991 as a collaborative project with the US Department of Energy. From 1993 to 1995 it continued as the ExternE Project, involving more then 40 European institutes from 9 countries, as well as scientists from the US.

Under the European Commission's Joule III Programme, this project has continued with three 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 describes the ExternE National Implementation Project for Finland. The objective of the ExternE National Implementation Project was to establish a comprehensive and comparable set of data on externalities of power generation for all EU member states and Norway.

The data in this report results from the application of ExternE-methodology as developed under Joule II. However, because our understanding of the impacts of environmental burdens on humans and nature is improving continuously, this methodology (or more precisely, the scientific inputs into the accounting framework) has been updated and further developed.

The National Implementation Project has generated a wide set of results, covering more than 60 cases, for 15 countries and 12 fuel chains. A wide range of generating options have been analysed, including fossil, nuclear and renewable technologies. Analysis takes account of the most important stages of the fuel chain, from (e.g.) extraction of fuel to disposal of waste material from the generating plant. In addition to the estimates of externalities made in the study, the project also offers a large database of physical and social data on the burdens and impacts of energy systems. The reference year for the calculated monetary values as well as for technology and other data was 1995 in this National Implementation.

The ExternE results form the most extensive externality dataset currently available. The original objective has been that the data could be used to look at a range of issues, including;

- internalisation of the external costs of energy
- optimisation of site selection processes
- cost benefit analysis of pollution abatement measures
- comparative assessment of certain energy systems

Such applications are illustrated by the case studies presented later in this report, and in other national implementation reports.

The Finnish National Implementation

Finland is situated in Northern Europe between the latitudes of 60° and 70°N and has common land boundaries with Russia, Norway and Sweden. The total land area of Finland is about 338 000 km^2 and its population is slightly more than 5 million. More than half of the Finnish population lives in the southern sixth of the country.

The most striking characteristics of the Finnish energy system are the importance of energy intensive industries, significant energy use for space heating due to the cold climate and long transport distances because of the sparse population. Consequently, the total per capita energy requirements are larger than in most other countries in Europe. The domestic energy resources are limited to hydro and wind power, nuclear power, peat, and renewable fuels. All of the oil, coal and natural gas requirements are covered by imports, and some electricity is also imported.

An important feature of the Finnish electricity generation system is the large share of combined heat and power production (CHP) in the overall electricity supply. Consequently, the average efficiency of fuel-based electricity generation in Finland is considerably higher than the average within the European Union.

The selected fuel cycles were coal, peat and wood-derived biomass, which together are responsible for about 40% of overall electricity generation in Finland and about 75% of the non-nuclear fuel-based generation. The ExternE methodology is aimed at the marginal approach, which means that the marginal impacts of *new* energy production capacity are of main interest. The selected fuel cycles therefore represent technology which would be utilised in power plants introduced at present and in the near future in Finland. In addition, gas-power plants could be built, if the increasing gas supply can be confirmed.

Methodology

The methodology used for the assessment of the externalities of the fuel cycles selected was that developed within the ExternE Project (EC, 1995). It is a bottom-up methodology, with a site-specific approach, i.e., it considers the effect of an additional fuel cycle located in a specific place.

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.

Comprehensiveness, at least to 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.

These characteristics should be present throughout the different stages of the methodology, namely: site and technology characterisation, identification of burdens and impacts, prioritisation of impacts, quantification, and economic valuation.

The ExternE Project uses the 'impact pathway' approach for assessment of the external impacts and associated costs resulting from the supply and use of energy. 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.

Quantification of impacts is achieved through the damage function, or 'impact pathway' approach. This is a series of logical steps tracing 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 impact. Several methods are available for this purpose, which will be adopted depending on the case.

Finnish criticism towards the methodology and its application

The proposed methodology also has serious limitations, which are discussed in a separate chapter and in the conclusions at the end of the report.

One weakness of the 'impact pathway' methodology is that the approach is 'atomistic' and might lack some vital information on the value of synergistic effects. The whole might be more than the aggregate of its parts.

The practical application of the methodology and presentation of the work could also be more transparent. The use of systematic taxonomic descriptions of the work would give a better overall picture of the structure of the research efforts, types of impacts included, methods of valuation, uncertainty of results and elements not included in the analyses. These descriptions are key elements when trying to improve the comparability of different externality studies. It is also important to notice that even the definition of external costs is not unambiguous, and that no clear distinction between external costs and other damages is actually made in the ExternE NI Project.

One problem is how the ExternE methodology should be applied in practice. In this NI study scientists or experts are responsible for several choices concerning priority impacts, valuation etc. It could be questioned whether these should be parts of a democratic decision-making process. The population which perceives these impacts and which bears their consequences also has a legitimate interest in the initial ranking of the impacts. It is possible that the adoption of monetary valuation might remove the key aspects of environmental decision-making from the sphere of public debate and place them in the hands of a small community of experts. Changing values cause additional problems: the externality may evolve with the passage of time, when the values and knowledge of society change.

The valuation of environmental impacts raises serious ethical and other important issues, which are outside the normal domain of welfare economics. It is not self-evident that different environmental impacts should be commensurate on a monetary scale. Some damage types do not have distinct property rights and are thus not ’tradable’. The divergence of WTP and WTA measures might also provide warning signals about valuation contexts, where the use of WTP and the application of cost-benefit frameworks to support decision-making is not admissible. It is typical of environmental non-market impacts that property rights have not been defined or may be conflicting and, in general, that the situation is not voluntary for an individual and that there might not be any substitutes for a ‘bad’ in question (e.g. risk of incurable illness). In general, the individual cannot influence the environmental risks via her/his own behaviour because these ‘public bads’ are indivisible in character. Furthermore, the person might not benefit from the production of incremental environmental burden. Thus, the whole context is different from the market conditions from which the underlying theories have been developed.

Utilitarian philosophy, which lies in the background of the goal of economic efficiency, may lead to a situation in which natural resources or services are allocated to the relatively small groups of people who gain the most benefit from them. When the basic measure of benefit is the willingness to pay (WTP) regardless of who causes the damages, the monetary valuation of environmental impacts allocates decision power and environmental services to the wealthiest groups of people.

It must be understood that the methodological framework itself is a ’measurement unit’, which observes reality from a certain fixed perspective. Thus the outcomes of analysis become meaningful only in the context of the applied methodology.

There are benefits and damages that cannot be assessed in economic terms and, on the other hand, it is possible that there is no empirical and ‘indisputable’ information available that could be applied in the decision-making, e.g. because of the future-orientation or time limits of these decisions. These deficiencies should be complemented by means of other valuation procedures.

Overview of the fuel cycles assessed

Coal fuel cycle

The power plant of the fuel cycle represents clean coal-firing technology. The Meri-Pori power plant was introduced into commercial operation in the beginning of 1994 as one of the world's cleanest and most efficient coal-fired power plants with a condensing turbine. Its pulverised coal boiler is Finland's largest boiler to date. The boiler is a once-through supercritical type with one reheat.

The flue gas cleaning ratio and the efficiency of this electricity generating power plant are notably better than those of other similar plants in Finland. The power station is equipped with the most modern gas cleaning facilities. The nitrogen oxides emissions formed in the boiler are reduced by 80% with the help of the low-NO_x burners and phased combustion and the catalytic denitrification system installed in the flue gas duct of the boiler. In the selective catalytic reduction (SCR) system, cleaning is based on ammonium injection and catalytic cells.

The plant runs on coal imported from non-EU countries. More than 2/3 of the coal is derived from Poland and Russia. In this National Implementation project it was assumed that all the coal is imported from Poland.

The major burdens of the coal fuel cycle are the atmospheric emissions of pollutants from the mining and power generation stages, liquid effluents and solid wastes from mining and power generation, and occupational accidents from the mining stage. The major air pollutants are SO_2, NO_x and CO_2. Total particulate (TSP) emissions (including fugitive dust) and CH_4 from mining are also significant.

The general selection of the priority impacts of the coal fuel is based on the results obtained in the earlier ExternE Project (EC, 1995). The most important of the impacts according to this valuation methodology seemed to be those caused by atmospheric emissions — especially from power generation — to human health. In addition, global warming is now considered as one of the major impacts.

Especially in the coal fuel chain considered here the liquid effluents from mining in Poland seem to have serious environmental impacts but their quantification could not be performed in this project. Occupational health impacts in Polish mines appear to be a very important part of the total health impacts.

The tool for calculating the dispersion of the primary pollutants TSP, SO_2 and NO_x — and the consequent impacts and damages — of power generation was the EcoSense 2.0 model. Because the computational grid of the model does not cover areas east of Finland, all the air emission impacts especially in the Russian areas nearby are missing in the basic model results. As an approximation the population of north-west Russia, about 8.4 million people were added to the most eastern gridcells of the model for sensitivity analysis of the results. After that the health damages of SO_2 and TSP were increased by 17% and the damages of NO_x by 31%.

The health damages were about 4 mECU/kWh, crop damages an order of magnitude lower and damages caused to building materials two orders of magnitude lower. The impacts on ecosystems are very difficult to assess, especially the long-term impacts. An even more difficult task is to assign any monetary valuation for this kind of damage. For all three fuel cycles, attempts were made for quantification of the *impacts only*. The ecosystem impacts were roughly quantified by the increase in land area where the critical load of acidity was exceeded. The impacts e.g. on biodiversity were not analysed. The ecosystems in Russia were not considered at all.

The damages due to the primary pollutants TSP, SO_2 and NO_x are all of the same order of magnitude. NO_x is however the most important of them due to its indirect impact on ozone formation.

Of the individual pollutants, CO_2 dominates due to its global warming (GW) impact. The damage estimates of global warming are considered to be very uncertain. Four different damage estimates (ECU / t CO_2) based on different assumptions and discount rates were presented by the ExternE Core Project. However, afterwards it has become clear that there might be some methodological confusion behind all the GW damage numbers applied in the NI project (end 1997). Using the lowest estimate for GW damage of the coal cycle, the damage is of the same order of magnitude as the damages of TSP, SO_2 and NO_x mentioned above, but using the highest estimate it is two orders of magnitude higher (about 120 mECU/kWh). Although the emissions of greenhouse gases and their global warming potentials (GWPs) are well known, knowledge of the true impacts and damages of GW is poor.

Peat fuel cycle

The Rauhalahti plant generating electricity, district heat and process steam is located in the Jyväskylä area in Central Finland. This CHP plant has a bubbling fluidised bed boiler. The main fuel is milled peat, but the new combustion technique enables the utilisation of wood fuels such as sawing waste, chips and bark as well as peat. Crushed coal and oil can also be burnt in the boiler. The efficiency of the plant is about 85%. In 1995 its overall fuel consumption was about 84% milled peat, 13% wood, 2% oil and 1% coal.

The flue gases of the power plant go through the electrostatic precipitator, which separates over 99% from the ash. The fluidised bed boiler reduces the nitrogen oxide emissions formed during combustion by over one third compared with the earlier pulverised boiler.

The peat is transported by trucks from peatlands in the vicinity of the plant and the average transportation distance is 80 km.

The major burdens of the peat fuel cycle are the atmospheric emissions of pollutants from the power generation stage. The major air pollutants are SO_2, NO_x and CO_2. Peat is considered a fossil fuel with global warming impacts. The emissions are calculated on a net basis. The emissions of the natural peatland are subtracted from the emissions of the calculated phases of peat fuel cycle. Natural peatlands are net sinks of carbon dioxide and sources of methane and nitrous oxide emissions. The amount of the emissions depends on the season.

The most important impacts of the peat fuel cycle are also those caused by atmospheric emissions. Liquid effluents from peatland ditching and peat production have environmental impacts such as eutrophication of the neighbouring water systems. Applying the ExternE methodology the human health impacts appear to dominate over those directed to other recipients. Global warming is probably the most important impact of the peat fuel cycle. Occupational health impacts of peat production were not considered in this study.

In the basic model results all the air emission impacts in Russia are missing. If this population were added to the model, it is reasonable to assume that the health damages would increase approximately in the same way as in the case of the coal fuel cycle. In the Externe methodology the impacts and damages are allocated to electricity and heat using the *exergy* principle, which appears to mistreat electricity generation in cogeneration plants in proportion to condensing plants. As a result of this most of the impacts/damages were assigned to electricity in the Rauhalahti case. In Finnish energy statistics the allocation is different and based on *energy* content.

The health damages were about 5 mECU/kWh, crop damages an order of magnitude lower and damages caused to building materials two orders of magnitude lower (as in the coal fuel cycle).

Considering the damages of the individual pollutants, the primary pollutants SO_2 and NO_x are of the same order of magnitude with each other, NO_x being the more important due to its indirect impact on ozone formation.

However, CO_2 dominates due to its global warming (GW) impact. The lowest GW damage estimate of the peat cycle is of the same order of magnitude as the damages of TSP, SO_2 and NO_x mentioned above, but using the highest estimate it is two orders of magnitude higher (about 140 mECU/kWh).

Biomass fuel cycle

The plant representing the biomass fuel cycle is a new combined heat and power generation (CHP) plant located in the town of Forssa. It began operation in autumn 1996, and is the first district heat and electricity producing plant of this size using solely wood biomass as fuel. The plant might represent a typical example of future energy technology in Finland that is environmentally more acceptable. The plant produces 95% of the district heat needed in the town and one third of the electricity supplied by the power company Forssan Energia to the power distribution network. Flue gases are cleaned with an electrostatic precipitator.

The fuel mix consists of saw dust, bark and wood waste. Almost 80% of the fuel is produced as a by-product from saw mills. Slightly more than 10% of the fuel comes directly from the forest and less than 10% consists of other kind of waste wood. The fuel chips coming directly from forest land are transported 0 — 50 km and the other wood waste fuels up to about 100 km.

The major burdens of the biomass fuel cycle are atmospheric emissions of pollutants from the power generation stage. The major air pollutants are SO_2, NO_x, TSP and N_2O. The CO_2 emissions from burning biomass are not taken into account as global warming impacts because the forestry is on a sustainable basis. However, the burning causes small emissions of N_2O, which is a powerful greenhouse gas. The moderate fossil CO_2 emissions are due to transport and production of the wood fuel and from oil which is used as an auxiliary fuel in the boiler.

The most important impacts of the biomass fuel cycle are those caused by atmospheric emissions from the power generation stage. The human health impacts appear to dominate over those directed to other recipients. Global warming impacts are small compared to those of the two other fuel cycles in this study.

The impacts of air borne pollutants on the Russian population are missing in the numerical results presented. As the stack at the coal plant is three times as high as at the biomass plant it can be assumed that the relative human health impacts in Russia are smaller for the biomass cycle than for the coal cycle. The impacts and damages of the biomass fuel cycle were allocated using the exergy principle as in the case of peat.

The health damages were about 6 mECU/kWh, crop damages an order of magnitude lower and damages caused to building materials two orders of magnitude lower (as in the coal and peat fuel cycles). The damages on ecosystems were not quantified, only some *impacts*.

Considering the damages of the individual pollutants, NO_x is the most important with damages an order of magnitude higher than those of TSP and SO_2. The indirect impact of NO_x in ozone formation increases its importance.

The highest GW damage estimate of the biomass fuel cycle (about 10 mECU/kWh) is of the same order of magnitude as the damage due to NO_x.

Aggregation

The Finnish electricity supply system includes almost 400 power stations with a total generation capacity of about 14 000 MW (1996). Small hydro or CHP plants are greatest in number, but the 10 largest plants (including four nuclear plant units) account for about 40% of the total capacity. Excluding the nuclear plants, the largest power plant is the Meri-Pori station, which was also included in this study.

An important feature of the Finnish system is the large share of CHP in the overall electricity supply. Consequently, the average efficiency of fuel-based electricity generation is considerably higher in Finland than the average within the European Union. The average efficiency for the year 1994 has been estimated at about 57% (Lehtilä et al. 1997).

The estimated damages of the three fuel cycles in the National Implementation Project were used as a basis for aggregation of the damages caused by the whole electricity generation sector. Two simple methods were applied. Only air borne pollutants (SO_2, NO_x, TSP and the greenhouse gases CO_2, N_2O and CH_4) were considered. All the aggregation results regarding the damages or external costs are uncertain, because the damage figures are very case-specific. The specific damages of sulphur, nitrogen and particulate emissions (ECU/t pollution) of the NI fuel cycles are dependent e.g. on the geographic location of the plant. Consequently, it is difficult to make any generalisations from these numbers to the whole electricity generation sector in Finland. The aggregation of total greenhouse gas emissions based on the individual fuel cycles is more realistic, and the impact of the GHGs is not dependent on the location of the power plant. On the other hand, the basic GHG damage estimates (ECU/t pollution), developed in the ExternE Project, are very uncertain**.**

Conclusion

In this study a large number of externalities for electricity generation were calculated based on the methodology and theoretical work of the earlier ExternE Project (EC, 1995a-f) and the Core Project (EC, 1998). The ExternE methodology is an attempt towards the integration of environmental impacts into energy economics. Here an additional object is the quantification of impacts in monetary terms so that the monetary results could be used in practical economic decision-making according to the discipline of neo-classical environmental economics. In the Finnish study human health related (especially mortal impacts) and global warming impacts dominate over those directed to other recipients and only these two appeared to be important in monetary terms. Their external costs could be high enough to affect decisions in energy policy. In the case of health impacts the monetary valuation of human life is decisive for the level of external costs. However, this valuation process is contradictory.

There are many uncertainties of diverse character in the results when using the impact pathway methodology of the ExternE. It is also possible that the 'atomistic' approach with distinct impact pathways might lose some vital information on the value of synergistic effects. The last part of the pathway is the monetary valuation of the impact, in which the uncertainties are also related to the subjective factors of the valuation process. There is a serious risk of misinterpretations when considering only the end results of the study — the total external costs or damages — without paying attention to the intermediate stages of the impact pathway (including the valuation criteria). A qualitative estimate of the externalities and impacts (and their uncertainties) before their monetary valuation is also important as well as an understanding of the methodological limitations. The scientists cannot be responsible for the valuation. It is doubtful how commensurable the impacts really are. It can be claimed that no purely analytical procedure can fulfil the role of a democratic political process in valuation.

An enhancement of the present National Implementation study would be a systematic and extensive taxonomic description of all the impact pathways. This would facilitate to see, for example, how far the impact pathways could be followed and which factors could not be assessed. The decision maker could then easier outline the problem and the limits of the results.

However, despite of the uncertainties and limitations of the methodology, it can be an effective tool in a comparison of similar kinds of fuel cycles, a new power plant and different pollution abatement technologies and different plant locations with each other. The relative differences in various impact factors might be more interesting and reliable than the absolute figures of external costs. Additional analysis concerning distribution of costs and benefits is still needed before putting traditional investment costs and costs estimated by means of the ExternE methodology side by side for optimisation purposes. This due to the fact that external costs are not based on voluntary behaviour of individuals like the other costs are. The strength of the 'bottom-up' approach is that the analysis is case-specific, taking into account all the concrete details of the fuel cycle technology under consideration.

1. Introduction

Economic development of the industrialised nations has been founded on continuing growth in energy production. The use of energy clearly provides enormous benefits to society. However, it is also linked to numerous environmental and social problems, such as the health effects of pollution of air, water and soil, ecological disturbance and species loss, and landscape damage. Such damages are referred to as external costs, as they have typically not been reflected in the market price of energy, or considered by energy planners, and consequently have tended to be ignored. Effective control of these ‘externalities’ while pursuing further growth in the use of energy services poses a serious and difficult problem. The European Commission has expressed its intent to respond to this challenge on several occasions; in the 5th Environmental Action Programme; the White Paper on Growth, Competitiveness and Employment; and the White Paper on Energy.

A variety of options are available for reducing externalities, ranging from the development of new technologies to the use of fiscal instruments, or the imposition of emission limits. The purpose of externalities research is to quantify damages in order to allow rational decisions to be made that weigh the benefits of actions to reduce externalities against the costs of doing so.

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. It was launched as the EC-US Fuel Cycles Study in 1991 as a collaborative project with the US Department of Energy. From 1993 to 1995 it continued as the ExternE Project, involving more then 40 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 chains. 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 (EC/OECD/IEA, 1995).

Under the European Commission’s Joule III Programme, this project has continued with three 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 the result of the ExternE National Implementation Project for Finland.

1.1. Objectives of the project

The objective of the ExternE National Implementation Project is to establish a comprehensive and comparable set of data on externalities of power generation for all EU member states and Norway. The tasks include;

- application of the ExternE methodology to the most important fuel chains for each country
- updating existing results as new data become available for refinement of methods
- aggregation of site- and technology-specific results to the national level

- for countries already involved in Joule II, data have been applied to policy questions, to indicate how these data could be fed into decision and policy making processes
- dissemination of results
- creation of a network of scientific institutes familiar with the ExternE methodology and data, and their application
- compilation of results in an EU-wide information system for the study.

The data in this report results from the application of ExternE-methodology as developed under Joule II. However, because our understanding of the impacts of environmental burdens on mankind and on nature is improving continuously, this methodology (or more precisely, the scientific inputs into the accounting framework) has been updated and further developed.

The National Implementation Project has generated a large set of comparable and validated results, covering more than 60 cases, for 15 countries and 12 fuel chains. A wide range of generating options has been analysed, including fossil, nuclear and renewable technologies. Analysis takes account of all stages of the fuel chain, from (e.g.) extraction of fuel to disposal of waste material from the generating plant. In addition to the estimates of externalities made in the study, the project also offers a large database of physical and social data on the burdens and impacts of energy systems.

The ExternE results form the most extensive externality dataset currently available. They can now be used to examine a range of issues, including;

- internalisation of the external costs of energy
- optimisation of site selection processes
- cost benefit analysis of pollution abatement measures
- comparative assessment of energy systems

Such applications are illustrated by the case studies presented later in this report, and in other national implementation reports.

1.2. Publications from the project

The current report is part of a larger set of publications, which commenced with the series of volumes published in 1995 (European Commission, 1995a-f). A further series of reports has been generated under the present study.

First, the current report covers the results of the national implementation for Finland and is published by Technical Research Centre of Finland (VTT). It contains all the details of the application of the methodology to the coal, peat and biomass fuel cycle cases and aggregation. Brief details of the methodology are provided in Chapter 2 of this report and in the Appendices; a more detailed review is provided in a separate report (European Commission, 1998a). A further report covers the development of estimates of global warming damages (European Commission, 1998b). The series of National Implementation Reports for the 15 countries involved are published in a third report (European Commission, 1998c).

In addition, further reports are to be published on the biomass and waste fuel chains, and on the application and further development of the ExternE methodology for the transport sector.

Preliminary results of the ExternE National Implementation for Finland have earlier been presented in some workshops (Pingoud and Pirilä, 1997; Pingoud, 1997).

This information can also be accessed through the ExternE website. The site is maintained at the Institute for Prospective Technological Studies, and is accessible through the Internet (*http://externe.jrc.es*). This website is the focal point for the latest news on the project, and hence will provide updates on the continuation of the ExternE Project.

1.3. Structure of this report

The structure of this report reflects the fact that it is part of a wider set of publications. In order to facilitate comparison of results, all ExternE National Implementation reports have the same structure and use the same way of presentation of fuel cycles, technologies and results of the analysis.

The common structure is especially important for the description of the methodology. Chapter 2 describes the general framework of the selected bottom-up methodology. The major inputs from different scientific disciplines into that framework (e.g. information on dose-response functions) are summarised in the methodological annexes to this report and are discussed at full length in the separate methodology publication (see above).

In order to improve readability, the main texts of the chapters dealing with the application to the different fuel cycles provide the overview of technology, fuel cycles, environmental burdens and the related externalities. More detailed information (e.g. results for a specific type of impact) is provided in the appendices.

1.4. The Finnish National Implementation

1.4.1. Description of the country

Finland, situated between 59° 30´ and 70° 5´ N and 19° 8´ and 31° 35´ E, is about 1100 km long in the north-south direction. The land boundary with Sweden in the west is almost 600 km long, with Norway in the north over 700 km long and with Russia in the east almost 1300 km long. The Baltic coast line is about 1100 km long (Figure 1.1). The land area is about 338 000 km^2.

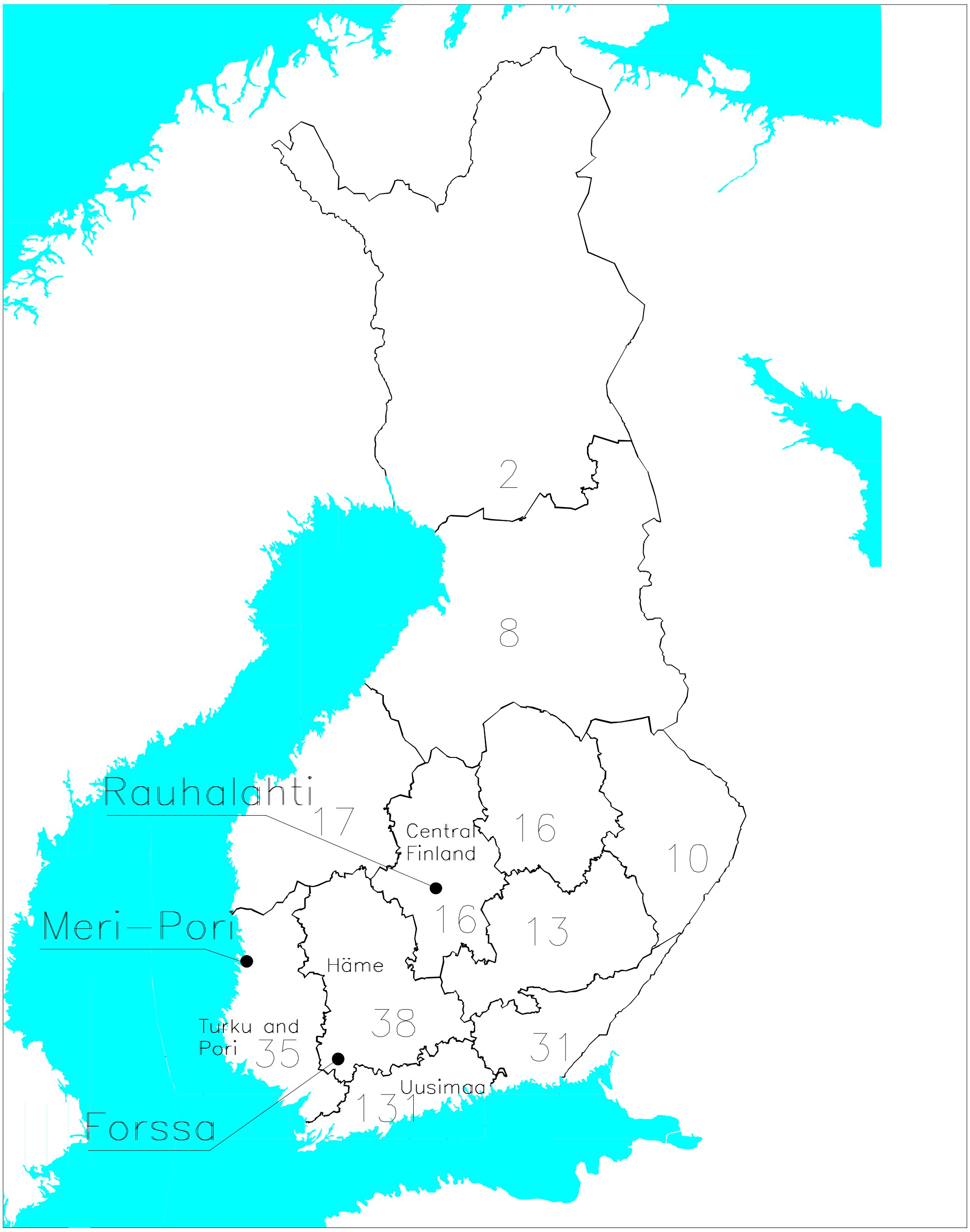


Figure 1.1 Finland: Locations of the power stations in Externe NI, and the inhabitants per km² of land area in the provinces[1)].

1) The number of provinces was reduced to five in September 1997. As statistical data is still better available for the previous situation the old figures and maps are used here.

The population of Finland is slightly more than 5 million and the average population density is less than 17 inhabitants per square kilometre, about one-twentieth of the more densely populated areas of Europe. Figure 1.1 also shows the population density in different provinces. More than half of Finnish population lives in the southern sixth of the country. The age distribution of the Finnish population is shown in Figure 1.2.

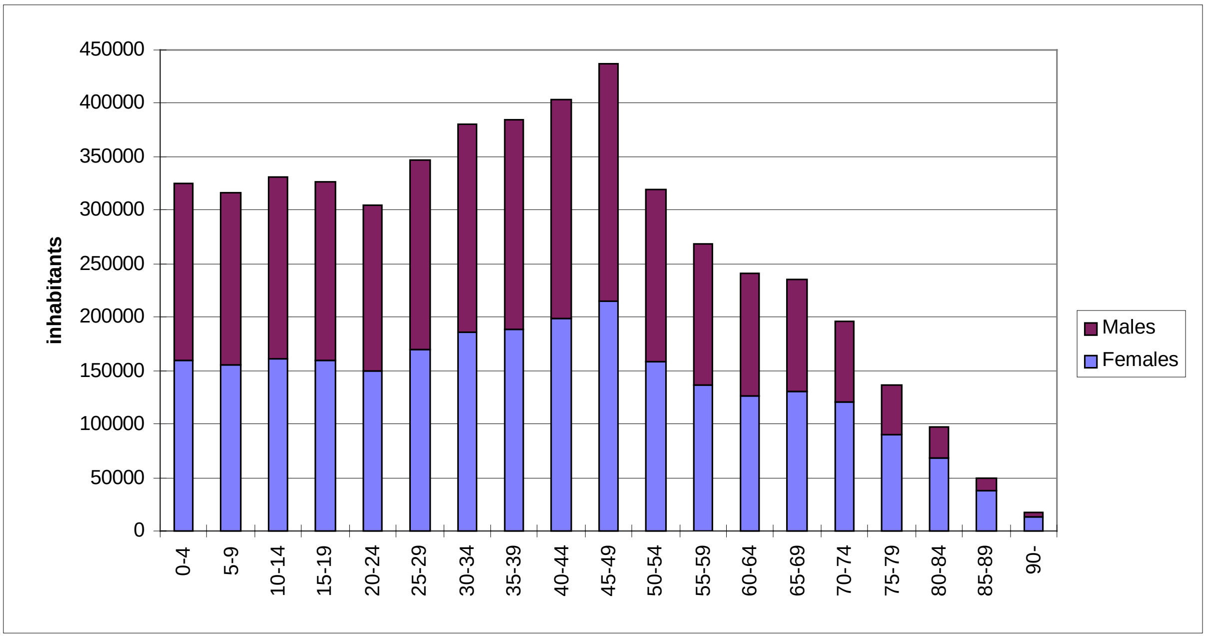


Figure 1.2 Population by age (Statistical yearbook of Finland, 1996).

About 10% of the area of Finland is covered by inland waters. Forest and other wooded land areas make up almost 70% of the land area. Finland forms a part of the boreal coniferous forest zone. In the southern part of the country, the conditions are ideal for coniferous forests. Towards the north, the climate becomes cooler and more humid. Mainly due to climate variation, forest increment varies significantly in different parts of the country. The dominant tree species are pine and spruce, dominating on 90% of the forest land.

Drainage of peatlands for forestry, agriculture, horticulture and fuel production was earlier an important issue in Finland. Peatlands were considered to have great potential for wood production. As a consequence about 60 000 km^2 of the peatlands in Finland have been drained.

The forest industries are very important for Finnish economy. The production of goods from sawn timber is more than 9 million m^3 and of paper and paperboard more than 10 Mt. The turnover in Finnish forest industries was more than 70 000 million FIM (about 12 000 MECU) in 1996 (Statistical Yearbook of Forestry, 1997).

The area of arable land in the 1990s has been about 25 000 km^2 or about 8% of total land area. Grain and grass are mainly cultivated but some sugar beet, potato and oil seeds are also cultivated. The utilisation of arable land in some rural districts 1992 is shown in Table 1.1. 5 000 – 10 000 km^2 of agricultural land will be set aside from conventional farming over the next few years.

Table 1.1 Characterisation of the region; use of arable land in four rural districts in 1992 (Maatilahallitus 1993).

Rural district	**Häme rural district**	**Turku rural district**	**Uusimaa rural district**	**Keski-Suomi rural district**
Use of arable land:				
- fallow	40 700 ha	66 100 ha	61 600 ha	24 100 ha
- uncultivated	4 000 ha	1 600 ha	6 500 ha	18 800 ha
- grass	31 200 ha	17 000 ha	27 000 ha	41 600 ha
- grain	78 000 ha	155 100 ha	121 600 ha	30 000 ha
- oil seed	8 000 ha	21 100 ha	15 200 ha	1 700 ha
- potato	2 600 ha	2 000 ha	1 600 ha	1 000 ha
- sugar beet	6 700 ha	10 700 ha	3 800 ha	
- peas	2 300 ha	7 500 ha	5 900 ha	900 ha
TOTAL	**173 500 ha**	**281 400 ha**	**243 200 ha**	**118 100 ha**

Picking of wild berries and mushrooms and hunting of deer and birds are additional forms of land use in forests and on mires. 5—10 million kg of wild berries and mushrooms are annually picked for sale (sale income for pickers 50—90 million FIM) in Finland. The catches of game had a value of 250 million FIM in 1992/93 (8 million kg of meat). These amounts of products taken directly from nature are small compared with normal meat production (annually 310 million kg 1995) and cultivating of crops for food purposes (annually about 5 000 million kg).

1.4.2. Overview of the Finnish energy sector

Some of the most characteristic features of the Finnish energy system are the importance of energy intensive industries (particularly the forest industries), significant energy use for space heating due to the harsh climate, and long transport distances because of the sparse population. Consequently, the total per capita energy requirements are larger than in most other countries in Europe. The domestic energy resources are limited to hydro and wind power, nuclear power, peat and renewable fuels. All of the oil, coal, and natural gas requirements are covered by imports, and some electricity is also imported.

The total primary energy requirements in Finland were about 1250 PJ in the years 1994 and 1995. This corresponds to about 240 GJ per capita, whereas the average total primary energy consumption in the European Union was about 150 GJ per capita in 1994. The structure of the total primary energy supply in 1994 is shown in Figure 1.3, both for Finland and for the EU average.

The most significant difference in the structure of consumption is in the use of biomass. Whereas the average share of biomass in the EU countries was only about 3%, in Finland the share was over 15% in 1994. Furthermore, peat is an important indigenous energy source, with a 5% share of the total energy consumption in 1994. On the other hand, the share of both natural gas and oil products is considerably smaller in Finland than the average for the EU.

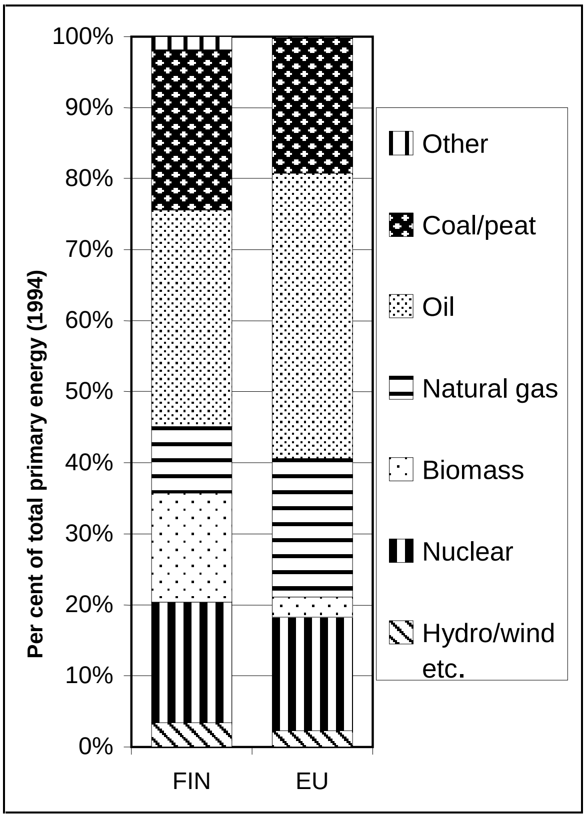


Figure 1.3 Structure of total primary energy consumption in Finland and EU.

Peat and wood-derived fuels in Finland

Finland has no fossil fuel resources, and the transport distances of imported fuels into Finland are generally rather long. Finland has large peat resources which can be utilised locally, since the peat-fired power plants are situated near the peatlands. Over the past two decades peat has become an energy source with a significant role in the Finnish energy economy. In 1995 the peat consumption was about 6% (1.86 Mtoe) of total energy. Power plants use about 66% of the peat. The major peat producers in Finland are Vapo Oy and Turveruukki Oy, accounting for 85% of the peat production.

In 1995 the area of peatlands used for energy production purposes was about 50 000 ha and total peat production was 24.7 million m^3 (2.1 Mtoe). In Finland there are 21 district heating peat boilers and 35 district heating boilers using peat and wood. There are one condensing power plant (150 MW) and about 30 industrial heat and power plants using peat and wood. The big peat power plants use milled peat and sod peat is used normally in the smaller plants. In 1995 the use of sod peat was 2 300 GWh and the use of milled peat was 16 290 GWh.

The total growing stock in Finnish forests is estimated at 1 887 million m^3 and its annual increment about 75 million m^3/a. When taking into account the whole forest biomass the growth is about 130 million m^3/a. Forest industries uses annually about 50 million m^3 of domestic roundwood. About 60% ends up as material in products, 40% (waste wood, bark and black liquor) is used for energy production. Total wood consumption has remained rather stable during the past 30 years, despite of a manifold increase in wood pulp production. This is mainly due to numerous structural changes, such as reduction in the non-industrial use of wood, reduction in round wood exports, increased use of industrial wood residues and the increased share of mechanical pulping and less wood-containing products. However, industrial wood consumption shows a strong upward trend, which means more waste wood for energy purposes as a by-product.

Today, when harvesting raw material from forests, about 29 million m^3 of forest residues are left in forests. The annual potential of harvestable forest residues is estimated at about 6.0—8.8 million m^3. This biomass potential is available if utilisation costs are low enough to make it competitive with traditional fuels.

Other potential biomass resources in Finland, not considered further here, are short rotation forestry and agro-biomass. Short rotation forestry has no important role in Finnish energy production as Finland already has an abundant potential of forest biomass and hence it is not particularly necessary to grow short rotation forestry. Among agricultural waste, straw could be used for energy production. The theoretical potential of cereal straw is about 0.7 Mtoe (28 PJ). About 20% of this could be utilised in energy production (VTT estimate). Among non-wood crops the greatest interest at present is in reed canary grass as 0.5—1 million ha of agricultural land will set-aside from conventional farming over the next few years. In 1995 reed canary grass was cultivated experimentally on an area of about 100 ha. However, the development of the energy crops sector is still at an early stage.

1.4.3. Justification of the selection of fuel cycles

An important objective in the National Implementation Project was to select fuel cycles which in some sense would be characteristic for the energy sector of Finland. In addition this would hopefully justify the aggregation or extrapolation of the case studies to the overall energy sector level. However, the ExternE methodology is aimed at the marginal approach, which means that the marginal impacts of *new* energy production capacity are of main interest. The selected fuel cycles therefore represent such technology, which would be utilised in power plants introduced at present and in the near future in Finland. All relevant future technologies in Finnish power generation could not be considered and for example the natural gas fuel cycle is not within the study.

The selected fuel cycles were coal, peat and wood derived biomass, which together account for about 40% of total electricity generation in Finland. Independently of this project, an assessment of the modification of hydro power regulation practices applying adapted ExternE methodology was performed by Imatran Voima Oy (IVO) (see Hongisto, 1997) and is not reported here.

A characteristic feature of Finnish energy sector is the significance of combined heat and power production (CHP) as described above, which was also taken into account in the selection.

In 1994 about 20% of the electricity consumed in Finland was generated by coal. The condensing power plant in Meri-Pori was chosen as a reference plant for the coal fuel cycle. It accounted for about 22% of coal fuelled generation in 1994. The Meri-Pori plant is one of the cleanest and most efficient coal-fired power stations in the world. However, coal is not one of the primary options in the future energy policy of Finland.

The power plants of the two other fuel cycles are of CHP type.

Peat is an indigenous energy resource in Finland representing 5% of the primary energy demand of Finland in 1995 and about 9% of the country's electricity was generated by peat. The Rauhalahti plant (located in the city of Jyväskylä), representing the peat cycle, is rather typical in Finnish energy production: a cogeneration plant producing electricity, district heat for the city and process steam for a paper mill.

The plant of the biomass cycle (located in the small city of Forssa) is new, introduced in autumn 1996. It is one of the first district heat and electricity producing plants, which can use wood biomass as sole source of fuel. Because Finland has in principle high potential in the utilisation of renewable energy resources, especially wood based biofuels, this plant might represent a typical example of future energy technology in Finland. The aggregation of this case to the whole bioenergy-based power production is still problematic. Most of Finnish bioenergy is generated in forest industries as a part of their production processes, and the results of the Forssa plant cannot be generalised directly to these processes.

1.4.4. Related national studies

The ExternE methodology has been applied in Finland also to wood biomass fuel cycle and in a some degree modified form to hydro power system by the energy corporation Imatran Voima Oy. The hydro case study deals with the impacts and damages of the modification of regulation practices of Oulujoki river and lake Oulujärvi. The analysed biomass burning plant, fuelled with sawing waste, bark and chips, is a small-scale combined heat and power plant located in Kuhmo, North-Eastern Finland. In addition to these, damage cost assessment methodology was demonstrated by means of existing literature for Loviisa nuclear power plant. These studies were reported in Finnish (Hongisto et al. 1998), and their results are briefly reviewed in English by Hongisto (1997). Ekono Energy Ltd. & Soil and Water Ltd. made the first nation wide top-down damage cost study for fossil fuel emissions (Otterström et al. 1994). Later on they used the developed methodology in a slightly modified form for local damage cost studies in Helsinki and Tampere regions (Otterström et al. 1995, Gynther et. al 1996). Attached to the latter study and in the subsequent study conducted in the Helsinki region Ekono Energy conducted two CVM-studies (i.e., Contingent Valuation Method) related to willingness to pay to avoid certain health effects. In these projects, direct comparisons with estimated damage costs and emission reduction costs were made by means of a cost-benefit framework. Environmental cost assessment methodology has also been applied to comparison of fuel options of wood chips, peat and coal (Ahonen et al. 1995) in the University of Oulu.

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. 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 in the framework of cost-effectiveness analysis 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. Although 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 and environmental impacts, 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.

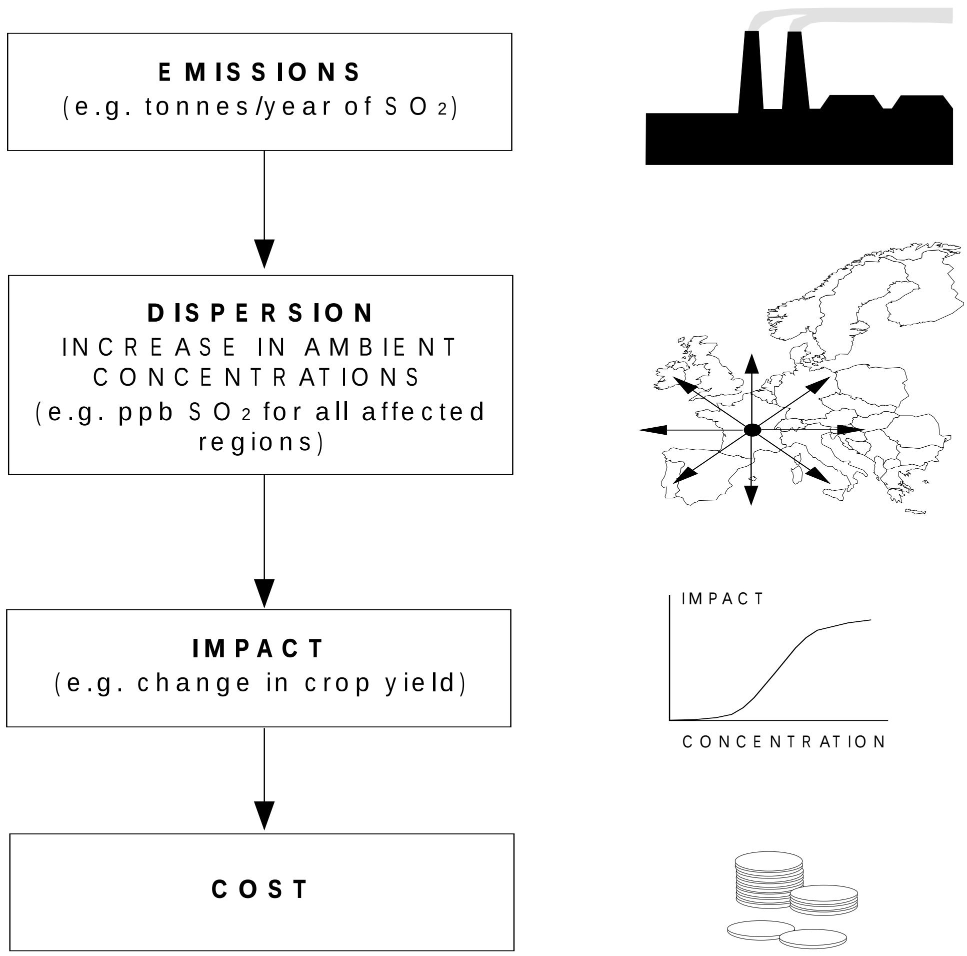


Figure 2.1 An illustration of the main steps of the impact pathways methodology applied to the consequences of pollutant emissions. Each step is analysed with detailed process models.

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 (as 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 of system boundaries and do not prioritise between different impacts. The ExternE analysts have to a large extent decided themselves whether certain stages of the fuel cycle, such as plant construction or fuel transportation, can be excluded. Such decisions are made from experience of the probable 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 emphasises 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.

Comprehensiveness, at least to 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 examines 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 to different fuel chains.

Certain impacts brought within these boundaries cannot be quantified at the present time, and hence the analysis is incomplete. This is not a problem peculiar to this type of analysis, but rather 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 demonstrate that this is the case — it cannot simply be assumed. Worth noting is the fact that variation in laws and other local conditions will lead to major differences between the importance of different stages in different parts of the world.

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 pollutants 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.

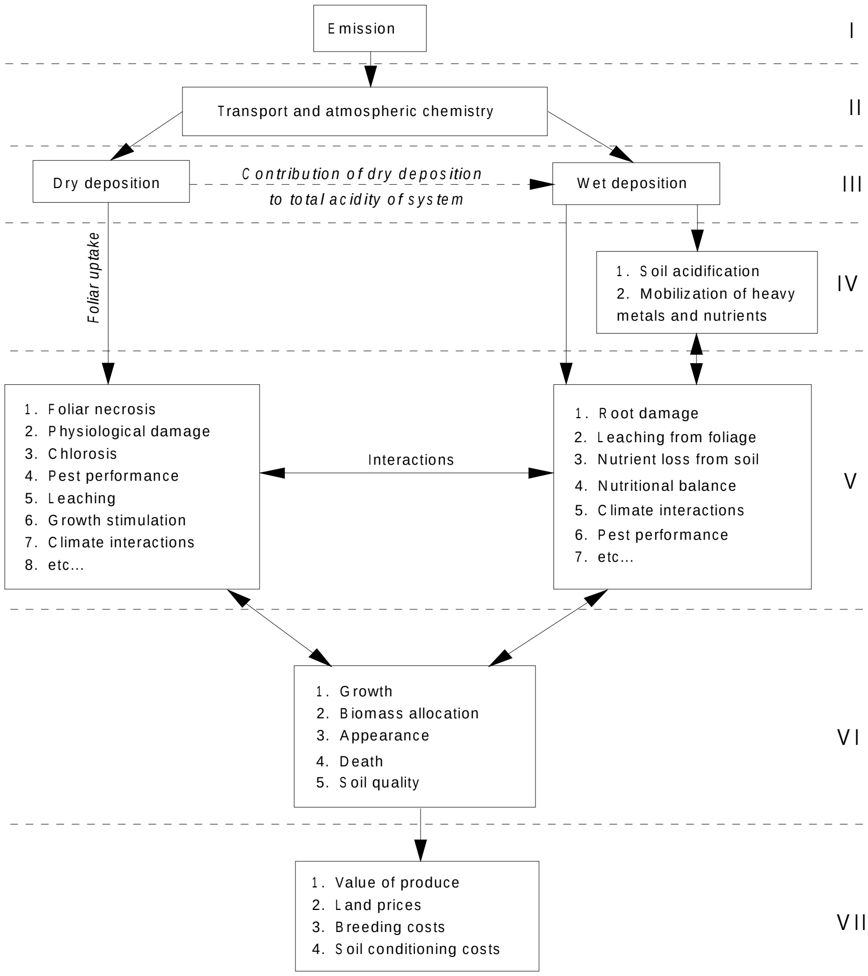


Figure 2.2 The impact pathway showing the series of linkages between emission of acidifying pollutants and ozone precursors and valuation of impacts on agricultural systems.

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 specific emissions 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.

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), VI (economic issues) and VII (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.

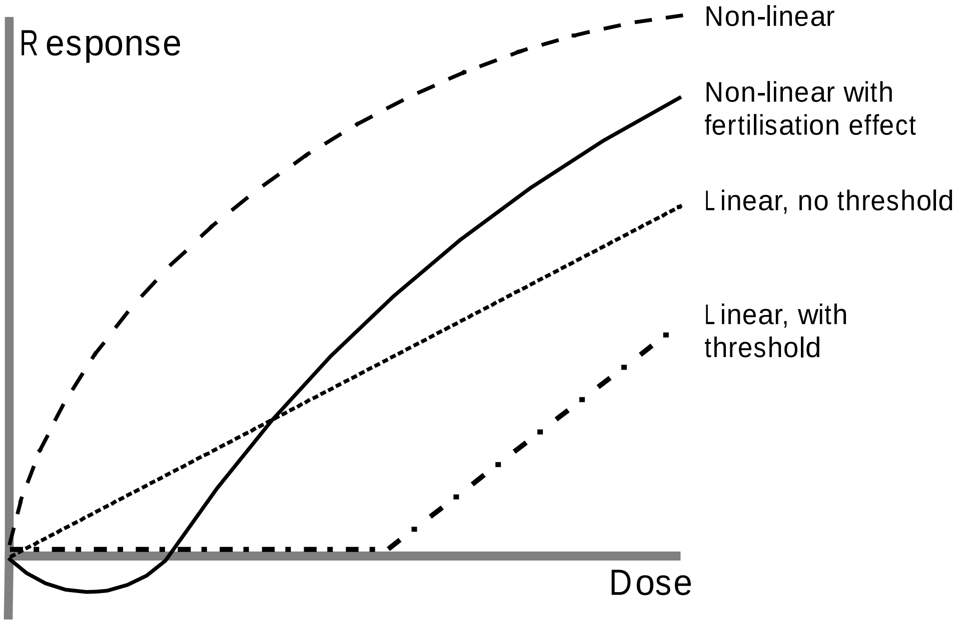


Figure 2.3 A variety of possible forms for dose-response functions.

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.

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 VI 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 VI). 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 VII, 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:

5. Occupational health effects;
6. Employment benefits and local economic effects;
7. Impacts of transmission lines on bird populations;
8. Damages to private goods (forestry, agriculture, water supply, ferry traffic);
9. 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.

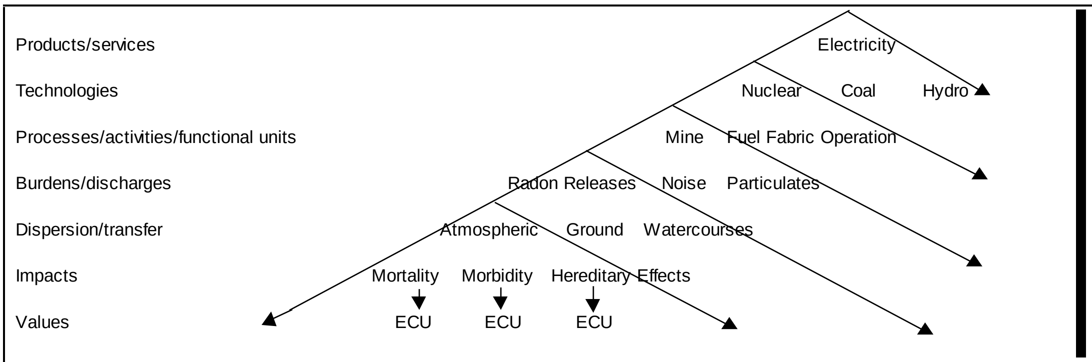
3. ExternE methodology — some notes by the Finnish National Team

In the following some basic features of the ExternE methodology presented in Chapter 2 are discussed and critical comments of the Finnish team are presented. The limitations of the assessment of external costs of power production were considered in the paper by Hongisto (1997) and in a report by Hongisto et al. (1998). The criticism of the ExternE methodology is based on these contributions.

In addition, some specific notes on modelling ecosystem impacts in ExternE are presented in chapter 3.4.

3.1. Impact pathway methodology

The general taxonomic structure of product-based externality study basically resembles the hierarchical pyramidal model in Box 3.1. The analysis of studies prepared using the 'bottom-up' methodology extends from the micro level burdens to the impacts by means of damage functions, whereas the analysis of 'top-down' studies runs from the macro level impacts into the burdens by dividing the total damages into parts. In principle, both research strategies should lead to the same conclusions.



Box 3.1 A hierarchical pyramidal taxonomic structure illustrating the conceptual levels of the impact pathway-damage function method. In principle, all levels are outcome levels, the data of which may be useful in decision making. Traditionally, the outcome of objective scientific examinations has been presented on the impact level. However, decision making requires clarification of the subjective values of impacts and answers to the question: what is important? In externality studies, the costs of the various types of impacts are added up into damage costs per produced commodity (Hongisto, 1997).

It is an important methodological issue that in the 'impact pathway' methodology the valuation is made on the level of end impacts and is then aggregated. However, it is possible that this ‘atomistic’ approach might lose some vital information on the value of synergistic effects, and thus the general impression might be more than the aggregate of its parts. (This problem may explain some of the magnitude differences between the outcomes of bottom-up and top-down studies.)

The value of each impact can be connected to the information elements located on an upper level in order to develop factors useful in decision-making. In practical applications, it is sensible to combine the categories and form compressed taxonomic presentations of the work (e.g. ORNL/RFF 1994, ESEERCO 1995 a, b). These are necessary tools for gaining a good overall picture of the structure of extensive research efforts, types of impacts included, methods of valuation, uncertainty of results and elements not included in the analyses. These descriptions are key elements when trying to improve the comparability of different externality studies.

It is important to note that it is often scientists or experts who select the priority impacts to be estimated. This approach has certain legitimacy in the definition of the initial importance of impacts on the environment. The population which perceives these impacts and which bears their consequences also has a legitimate interest in the initial ranking of these impacts. It would therefore be desirable that they could also participate in the selection of priority impacts. In this way the choice process would become more democratic (ENVECO, 1997). It is possible that the adoption of monetary valuation might remove the key aspects of environmental decision making from the sphere of public debate and place it in the hands of a small community of experts.

3.2. Problems of valuation

The valuation principles in the ExternE methodology are discussed in section 2.4.6 and in Appendix VI. Here some comments on the involved valuation problems are given.

One problem is changing values, also considered in Appendix VI: the externality may evolve with the passage of time, as the values and knowledge of society change. It is important to notice that the definition of external costs is not unambiguous, but appears in various forms in the practical applications made in the energy sector. Externalities cover very different types of costs in different studies conducted in the 1990s in the USA and Europe. However, in most cases the externalities mean environmental damage costs, without taking a stand on whether the value of the damage has been considered in previous decisions or whether the magnitude of damage would require some further action to be taken. In terms of economic decision-making, not all environmental damage costs are defined as relevant externalities; some of them are called residual damages. In terms of practical decisions on energy issues, an accurate division of the environmental damage costs into residual damages and economically relevant externalities constitutes a very theoretical and subtle line because of the great uncertainty of damage costs, the number of administrative controls and economic instruments, eventual overlapping, the range of liabilities of institutions affecting decision-making, and the complexity of representative decisions. The roles of institutions, laws and the democratic decision-making process are generally disregarded if the analysis is based purely on neo-classical economic analysis.

The monetary valuation of impacts aims at internalising the damage to the decision-making process by making possible the compensation of residual damages in monetary units, at least in principle. In the area of residual damages, ‘trading’ is thus realised by means of environmental impacts, which means that the impacts are not prevented from emerging but are compensated instead. It is thus of extreme importance that the impacts of new projects are valued correctly, i.e. the level of compensation is commensurate with the additional damage caused. The empirical evidence is thus very important and cannot be replaced by means of selected literature data without losing overall credibility.

The value-related decisions are often made implicitly and almost subconsciously (in the choice of method, data etc.), thus making it difficult to discuss the values and the implications of different standpoints (Finnveden, 1997). The valuation of environmental impacts raises serious ethical and other important issues, which are outside the normal domain of welfare economics. If valuation measures are used for the study of many public policy questions instead of the democratic decision-making process, it is of utmost importance that the measure of value is generally accepted. In view of many environmental impacts, their commensurability on a monetary scale should not be treated as self-evident, due to problems related to the monetary valuation of damage types, which do not have distinct property rights and are thus not ’tradable’.

Some of the ethical problems of monetary valuation are associated with assumptions, which are in turn associated with property rights, and some are linked to underlying utilitarian premises. Without property rights, exchange is not necessary (or possible) and thus the valuation of the object in monetary terms can be considered as more or less meaningless. However, this does not mean that the object is valueless — the object is only non-exchangeable in its character.

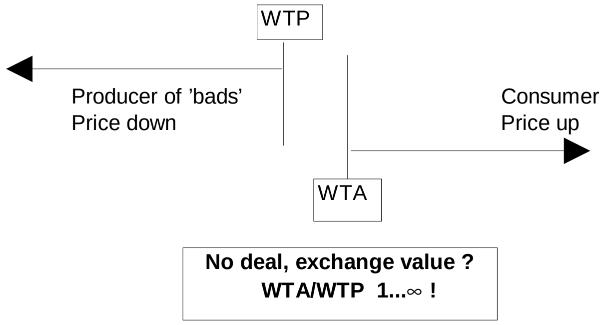
In market conditions, exchanges of goods and services (or some property rights) are made on the basis of voluntary behaviour and choices. The consumer and producer of the goods or services decide when the price or ‘exchange value’ is right and commensurate with the goods or services. Otherwise, the exchange of some property rights (by means of monetary units) does not occur, the exchange value of the goods cannot be detected and no commensurability in monetary units exists. In these cases there is no overlapping area between consumer willingness to pay (WTP) and producer willingness to accept compensation (WTA). However, all is not for sale and everything cannot be bought from the markets.

It is known that there is a divergence between the WTP measures when individuals buy and the WTA measures when they sell the same object. Hanemann (1991) demonstrated that the divergence can range from zero to infinity, depending on the degree of substitution between goods. One should only expect convergence of WTP and WTA measures when the goods in question have a very close substitute. A value divergence will exist and expand as the degree of substitution decreases. It has been shown that for non-market goods with imperfect substitutes, the divergence of WTP and WTA value measures is persistent even with repeated market participation and full information on the nature of the goods (Shogren et al., 1994).

It is typical of environmental non-market impacts that property rights have not been defined or may be conflicting and, in general, that the situation is not voluntary for an individual and that there might not be any substitutes for a ‘bad’ in question (e.g. risk of incurable illness). In general, the individual cannot influence the environmental risks via her/his own behaviour because these ‘public bads’ are indivisible in character. Furthermore, the person might not benefit from the production of incremental environmental burden. Thus, the whole context is different from the market conditions from which the underlying theories have been developed. The finding of divergence of WTA and WTP, in some cases from zero to infinity, and the context-dependent reasons behind this, form an important issue from the point of view of the monetary valuation of environmental impacts in general.

Applying the economic approach to evaluate non-market environmental impacts, it is necessary in the lack of real markets to resort to WTP or to WTA, as expressed in the CVM studies, as a measure of individual preferences. The person carrying out the CVM studies is forced to make some background assumptions on the property rights of the subject to be valued and on the monetary substitutability of non-market damage, i.e. ‘necessity of exchange’, when deciding whether to select WTP or WTA as the measure of value and how to handle any refusals to answer or, possibly, ‘unreasonably’ high compensation requirements. WTP is constrained by income but WTA is not. Problems might emerge due to ‘infinite’ compensation requirements, which might reflect the noncommensurable or irreplaceable nature of the assessed ‘bads’. Which measure is correct? The answer depends on the context and especially on property rights. WTA is the correct measure when the individual has a property right and is being asked to forego a benefit or to accept a loss. However, when estimating externalities of additional ‘private’ power projects in the context of liberalised electricity markets, which might damage the environment, in most recent studies the damages have been estimated erroneously by means of WTP measures, which have been transferred from a very different context (mostly from traffic safety studies to the energy sector) to new applications disregarding their context specificity and the multidimensionality of risks.

It could be questioned what makes the use of WTP at low damage levels acceptable when it is clearly unacceptable on high level risks of death? The reliance on WTP measures in current externality approaches is doubtful, because it implicitly means that ’property rights of environmental damages’ are interpreted as belonging to the source of damage in all cases disregarding the context of how costs and benefits of production are distributed. However, if the choice is made inversely, WTA measures — e.g. those attached to serious risks — can increase towards infinity if they are perceived as irreplaceable. Thus the use of WTA instead of WTP in certain contexts would reduce the ethical problems of monetary valuation, and on the other hand, its overall applicability in environmental decision-making. The divergence of WTP and WTA measures could provide a warning signal about the valuation contexts, where the use of WTP and the application of cost-benefit framework to support decision-making are not admissible. However, the rights of future generations and the evaluation of the fauna and flora are still controversial. What do we know about the preferences of future generations in terms of WTA?



Box 3.2 For most non-market environmental impacts property rights are not set. However, the choice of WTP or WTA includes assumptions on the property rights of the subject to be valued and on the monetary substitutability of damage. The latter is self-evident in the case of voluntary trade but not always in the case of external impacts or 'public bads' if there is no overlapping area between WTP and WTA. The gap between WTP and WTA measures increases when the degree of substitution of the valuated issues decreases (e.g. when the risk of death increases). The problem illustrated in this figure might have significant impact on the monetary value of environmental impacts and on the applicability of CVM studies to support decision-making.

It can be concluded that the valuation of environmental damage in monetary terms implies acceptance of a monetary compensation for damage; this in turn should lead to the damaged group of people being allowed to decide on the commensurability of the damage and its monetary value as well as on the accepted level. Otherwise the question of whose values should be used is left in the hands of analysts, and damage costs of 'public bads' can be considered to be born involuntarily and cannot be compared with other market prices of 'private goods' born by voluntary exchange, choices and behaviour. This is something very distant compared to the current practise of utilisation of scientifically transferred 'universal' values and the embedded assumption of general monetary substitutability of all environmental impacts.

Ultimately, the commensurability in monetary units is tested in the case of residual damages, which remain after optimisation of control measures in terms of economic efficiency. If WTP were used as a measure of value, i.e. property rights were interpreted as belonging to the source of damage, and decisions were made purely on the basis of neo-classical economics disregarding the distribution of costs and benefits, nothing would guarantee e.g. the health and safety of individuals in the poorer parts of the world (due to climate change, hazardous wastes etc.). Utilitarian philosophy, which lies in the background of the goal of economic efficiency, may lead to a situation in which natural resources or services are allocated to the relatively small groups of people who gain the most benefit from them. When the basic measure of benefit is the willingness to pay (WTP) regardless of who causes the damages, the monetary valuation of environmental impacts allocates decision power and environmental services to the wealthiest groups of people.

At the policy level it should be taken into account that the public responsibility for potential consequences of major infrastructural decisions cannot be transferred to the analysts, who made the studies, nor to the programs, frameworks or theories applied. There are benefits and damages that cannot be assessed in economic terms, and, on the other hand, it is possible that there is no empirical and 'indisputable' information available that could be applied in the decision-making, e.g. because of the future-orientation or time limits of these decisions. These deficiencies should be complemented by means of other valuation procedures.

3.3. General remarks on the methodology and its practical application

The ExternE-methodology is based on damage cost valuation. Control costs are seen to belong only to one side of the cost-benefit-equation as an element which does not fit the background assumptions related to individualistic valuation. The valuation of climate change, accident risks and depletion of biodiversity and resource base by means of damage-cost methods has proved to be extremely difficult. However, related to these issues high control costs and significant public attention show that these questions are socially important even if they cannot be valuated with any accuracy by means of damage cost methodology. In the context of representative social institutions and political decision-making procedures, cost-effectiveness analysis based on control costs is often applied to support decisions instead of the application of a cost-benefit-analysis (CBA) framework based on damage costs. Thus control costs form an important element to support decision-making under uncertain conditions. According to the principles of representative democracy, control costs reflect to a greater or lesser degree the public preferences associated with the perceived environmental damages. However, it must be emphasised that the transfer of all cost components and other sub-elements of studies from one context to another is doubtful and involves the risk of omission of cultural and site-dependent features are omitted in the decision-making process.

The efforts of the ExternE Project and the US-DOE have demonstrated the complexity of full-scale fuel-cycle analyses. Due to the scale of the reports, the communication of framework-dependent results and other important information becomes a problem. One way to improve the utility of data at hand is to describe the architecture of the study by means of systematic taxonomic descriptions. In these tables it can easily been shown which processes of the fuel cycle, which burdens and impacts have been incorporated into the total results of the analysis and how comprehensive the conducted analysis of various burdens actually is. A taxonomic description in the form presented e.g in ORNL/RFF (1994) improves the intelligibility and comparability of different studies.

It is important for the decision-makers that they could determine which issues are covered by the analysis and which are not, and are thus subjects for further (alternative) assessment and valuation procedures. Without these descriptions, and combined with the lack of financial resources of the study, the lack of reliable scientific knowledge on potential (possibly new) impacts and the lack of empirical damage cost valuation studies, there is an obvious risk that application of the CBA framework might lead to the conclusions, which underestimate the value of potential impacts. The choices and assumptions made in central issues — concerning the application of a CBA-framework and especially the monetary valuation but also the impact assessment phase — have such a significant influence on the results that they should always be reported transparently and completely in connection with the obtained results. Otherwise, there is an obvious danger that decision-makers may be misled.

In future studies more emphasis should be placed on understanding the allocation problems related to recycling of materials and production of several energy products (CHP) simultaneously. In addition to these, principally similar allocation problems also associate with the formation of exposure-response functions by means of epidemiological and ecological studies and value functions by means of CVM-studies. There are numerous circumstances in which synergistic multiple effects might exist and weaken the reliability of the results. Site specificity and non-linearities of impacts and values are a challenge for science and especially for the practical applications.

When assessing how ’real’ the estimated values are, it must be understood that the methodological framework itself is a ’measurement unit’, which observes reality from a certain fixed perspective. Thus the outcomes of analysis become meaningful only in the context of the applied methodology. Multi-dimensionality of environmental impacts enables various interpretations about these effects from the perspectives of various scientific disciplines. According to the pluralistic idea, the analysis of ’exchange values’ can be seen only as one perspective to these issues among several others.

Several practical problem areas can be detected, which may lead to non-commensurability and rule out the use of results from decision-making processes. This might occur even if the studies have been conducted technically in an excellent way. Some key problem areas are:

1. Ethical and equity issues related to the monetary valuation of environmental impacts,

2. Disagreement concerning goals on a policy level,

3. Inadequate illustration of overall uncertainties,

4. Lack of empirical data and reliance on transferred elements or chosen ‘blocks’ of previous studies,

5. Unknown or unequal boundaries of the analysis (place, time, system, impacts),

6. Lack of analysis of certain types of impacts perceived as important (risks, effects on ecosystems etc.),

7. Possibility of institutional capture by choices of methods and the way in which the analysis is conducted,

8. Time delay between source data in relation to actual decision-making situations, etc., and

9. Uneven level of knowledge and accessibility of data associated with various impact types.

Basically, commensurability between the outcomes of externalities (or damage costs) of different fuel cycles and different studies and between various impacts in one study is a matter of faith and depends on the level at which the comparisons are made. Ultimately, the commensurability depends on the valuation methods applied. Disputes over the valuation methods mainly centre around the utility and accuracy of different types of evidence (OTA 1994). For example, the market, hedonic and travel cost methods draw their information from consumer choices, the contingent valuation (CVM) from interrogation studies, and the control cost method from the decisions of democratically elected politicians. It can be questioned, whether the obtained results are commensurable.

If the CBA-framework is utilised in decision-making, the evolution of cost components on both sides of the cost-benefit equation should be followed even after the active decision-making situation. Nevertheless, the time delay between the source information and the actual decision-making situation might become a problem. For example, new technological innovations tend to decrease the presumed control costs and thus 'leave room' for stricter control and norms. On the other hand, this is an example of how technological expectations related to new innovations might 'ingrain' into the policy process. This emphasises the need to improve the flexibility and interactivity of the conventional CBA-based decision-making process.

CBA-based decision-making is insusceptible to problems related to potential new burdens such as new chemicals. It is based on already existing information. Thus potential hazards are valued by means of the principle 'everything is safe until otherwise indicated'. One important question is, how to balance the cost of provision of adequate knowledge for the evaluation of an 'unreasonable risk'. Who should pay the potentially high costs?

3.4. Evaluation of ecosystem impacts in ExternE

3.4.1. Forest growth assessment

An extensive search for quantitative dose-response functions within the ExternE Project was carried out to describe monetary benefits for reductions of air pollutant emissions (EC 1995b). This review on the applicability of ecosystem effect valuation to Finland focuses on the forest growth response to acid deposition.

Various relationships with acid deposition, exceedance of critical loads and the critical loads themselves were explored (Kuylenstierna and Chadwick, 1994). The best correlation was found between the country-average fraction of forest area in defoliation classes 2–4 (corresponding to needle loss above 25%) and the average exceedance of forest critical loads (in meq m^{-2} a^{-1}). To make the defoliation value usable for possible economic assessment, it was related to growth responses using the results of a Swedish study (Söderberg, 1991). The data suggest a relationship between the degree of defoliation and a decrease in growth increment in the sampled trees, representing both Scots pine and Norway spruce in different areas of Sweden.

The acid deposition of 1980 (sulphur deposition minus total alkaline base cation deposition) was compared with average defoliation data from 1987-90. There have been large reductions in sulphur emissions during the 1980s and 1990s (e.g. Barrett and Berge, 1996), and a decreasing trend has also been shown for the base cation deposition (Hedin et al., 1994). The use of depositions from additional years or a cumulative dose could reduce some uncertainties in the correlations.

The correlation of defoliation with critical load exceedances assumes that indirect acidification effects via soil are largely reflected in defoliation. The area of severely defoliated forest in each country, which could have been used as a weighing factor, did not enter the correlations. At zero critical load exceedance the regression predicts that 13% of the forest area falls in defoliation classes 2-4, which thus cannot be explained by critical load exceedances. The fraction that can be explained, according to the regression figure, happens to be always less than the aforementioned amount.

The data used to map the ecosystem sensitivity is based on the work by the Stockholm Environment Institute. These critical load data are different from those compiled by national focal centres for use in preparing UN-ECE emission reduction protocols (UBA, 1996; Posch et al., 1997). The use of critical loads in assessing observed damage should differentiate steady-state and dynamic aspects. Not all recipients have lost all their buffering capacity and thus they do not currently manifest observable detrimental effects, although in some time scale the continuing excess load will evidently lead to such a state. Therefore, defoliation figures together with data on critical load exceedances should be used with extreme care recognising the importance of the dynamic aspect.

Potential uncertainties in defoliation results are the country-specific variation in monitoring methods and practices and the spatial defoliation variation within countries. The relationships between defoliation and air pollution have been explored in Finnish defoliation monitoring programme reports (e.g. Jukola-Sulonen, 1990). It has been noted though that gaseous air pollutants have had some local effects on vegetation. No clear correlations of sulphur and total nitrogen loads with defoliation have been found, even when stand age was eliminated (Müller-Edzards et al., 1997). Increased defoliation was detected at some young pine stands in nutrient-poor soils with high levels of weatherable aluminium in western Finland with moderately high sulphur deposition. In general, defoliation does not correlate with the chemical soil acidity parameters. Studies on long-term forest growth and its potential responses to natural and anthropogenic factors in Finland were examined by Nöjd (1990). The results revealed no long-time adverse changes or forest growth decrease due to air pollutant load, although relative variations in the susceptibility of different soils and tree species have been pointed out.

The relationship of defoliation to growth responses is based on a study of the Swedish data (Söderberg, 1991). The results show that increasing tree defoliation is reflected in diminishing growth rate, although not consistently for all regions in Sweden. The study could not resolve the possible roles of climate and pollutant load in defoliation severity and consequently tree growth responses. The data cover only single tree responses to environmental stress factors and the response of an entire forest stand is not depicted by the approach. Therefore, the use of these results should be treated as indicative only. Similar results to those of Söderberg (1991) were found in the data from 8th National Forest Inventory in southern Finland in 1986-87, correlating defoliation severity with growth decrease detected from tree ring analysis (Nöjd, 1990). Trees with needle loss of 41-50% had, on average, lost 20% of their diameter increment. In the conclusions it was clearly stated that the use of the single tree growth decline is very unreliable in estimating possible growth reductions at the stand level.

3.4.2. Discussion on other ecosystem effects

The effects of ground level ozone have been preliminarily estimated in Finland using the criteria proposed by an UN-ECE workshop (Kärenlampi and Skärby, 1996). There was no risk to forests according to the suggested criteria, but the critical level for crops was exceeded at many of the stations. The models on ozone exposure at a grid of 150 km $\times$ 150 km over Europe do not indicate potential vegetation effects in Finland (Barrett and Berge, 1996), although discrepancies between measured and modelled values exist (Lövblad et al., 1996).

The data on critical loads of acidity for lakes were not used in the EcoSense model. The small headwater lakes sensitive to acidification have been regarded as important ecosystems in Finland in addition to the forest soils. The exceedances of critical loads for lakes could be included as indicator values in the assessment in the same way as for forest soils, since both ecosystems are an integral part of the national critical load database. No natural terrestrial ecosystems are currently included in the critical load mapping programme in Finland.

3.4.3. Comparison between EcoSense and national modelling results

The effects of additional power plant emissions on critical load exceedances were calculated with national models and databases including both lake and forest soil ecosystems (Kämäri et al., 1991; Posch et al., 1997; Syri et al., 1997). The additional increases in total ecosystem area where the critical loads for acid deposition are exceeded were 0 km^2 around Forssa, Kiimassuo, 14 km^2 around Rauhalahti and 29 km^2 around Meri-Pori. The increase is largely attributed to the exceedance on forest soils. The soil data were mapped on a more even grid than the nationally representative cluster-sampled lake survey data, which in this case left the areas near plants practically without lake calculation points. The results of the EcoSense model showed increases in severely defoliated forest area of 0.3, 4 and 2 km^2, respectively, which approximately correspond to the values resulting from the national model. The outcome demonstrates the effects of different input data and the sensitivity of the approach due to only a small increase in dose caused by an increment of a single plant.

3.4.4. Conclusions

The evaluation under ExternE on ecological effects has resulted in a compilation of potential impacts of several pollutants on a variety of ecosystems. In the evaluation of effects of the soil-mediated effects of acidifying air pollutants, a relationship between critical load exceedance, defoliation and consequently decreases in growth increment was suggested. The presentation of the hypothesis discussed most potential caveats of the approach, although no range for the uncertainty of the results was given. Part of the defoliation was explained by sulphur-related acid deposition exceedance critical loads when country-averaged defoliation values were examined. The extension of the relationship between tree defoliation and decrease in tree diameter growth increment to forest response and potential timber losses remains an uncertainty in the assessment. Currently, studies carried out in Finland do not support within-country correlation between acid deposition, critical loads and defoliation. In relation to the acidification problem, the inclusion of lakes sensitive to acidification would result in an evaluation more consistent with the critical load mapping activities. Ozone-related effects may have local importance difficult to depict using models with coarse spatial resolution. The comparison of EcoSense results to national critical load assessment illustrates the differences in input data and the uncertainty arising from applying small incremental doses. The results and possible monetary valuation of ecological effects must be carefully evaluated due to uncertainties in current dose-response functions and their applicability to different countries and ecosystems.

4. Coal fuel cycle

4.1. Definition of the fuel cycle

A detailed definition of the Finnish coal fuel cycle is given in Appendix VIII, where the burdens, impact and damage figures of the fuel cycle are also listed.

4.1.1. Power generation stage

The Meri-Pori power plant was introduced into commercial operation in the beginning of 1994 as one of the world's cleanest and most efficient coal-fired power plants with a condensing turbine. Its pulverised coal boiler is Finland's largest power plant boiler to date. The boiler is of the once-through supercritical type with one reheat.

The flue gas cleaning ratio and the efficiency of this electricity generating power plant is significantly better than those of other similar plants in Finland (Imatran Voima Oy, publication year not given). The power station is equipped with the most modern gas cleaning facilities. The nitrogen oxides emissions formed in the boiler are reduced by 80% with the help of the low-NO_x burners and phased combustion and the catalytic denitrification system installed in the flue gas duct of the boiler. In the selective catalytic reduction (SCR) system, cleaning is based on ammonium injection and catalytic cells.

Particles are separated from flue gas by an over 99.5% efficient electrostatic precipitator located before the wet flue gas desulphurisation plant. About 90% of the sulphur dioxide is removed by the desulphurisation plant. Flue gas is passed through a scrubber with circulating limestone slurry. Sulphur dioxide reacts with slurry and the end product is gypsum.

Purified flue gases are exhausted through two 150-metre stacks.

Table 4.1 Specifications of the fuel.

Country	Moisture content (%)	Ash content (%)	Calorific value (MJ/kg)	Sulphur content (%)	Carbon content (%)
Poland	8—9	12—14	25.6	0.6—0.7	74
Russia	7—8	16—18	24.7	0.8—1.0	71

The plant runs on coal imported from non-EU countries. Most of this coal is derived from Poland and Russia (more than 2/3) and the remainder from other countries including the Republic of South Africa, Australia and Indonesia. The specifications of Polish and Russian coal are given in Table 4.1. In this National Implementation Project it was assumed that all the coal is imported from Poland.

4.1.2. Upstream fuel-cycle operations

No reference mine in Poland or Russia could be identified because of lack of information, but underground mining in Poland is described in general. All Poland's hard coal is produced from underground mines. Most of the coal comes from the Upper Silesian coalfield (98% of the total output in 1990) (Walker, 1990). The Polish hard coal mining has achieved a rate of mechanised mining of 98.3% along with complete electrification of all the basic technological processes (Polish Coal, 1991). Coal extraction is exclusively by longwall mining and mainly employs total caving.

Almost all Polish coal passes through a washery or preparation plant. The most common preparation technique is dense medium (typically magnetic) separation. However, jigs are also used. (Doyle, 1989) In 1988 about 24% of the run-of-mine output was separated at coal preparation plants.

About 95% of Poland's inland coal is moved by rail. Coal imported to Finland is transported from the Upper Silesia basin to Gdansk by rail. The unit trains operating on the Katowice to North Port line have a net capacity of 2600 tonnes (forty-four 60 tonne wagons) (Doyle,1989). Poland's ports can handle only medium-sized vessels (up to 100,000 dwt) (Mannini, 1989).

The fuel is delivered to the power station by ship. The entrance channel leading to the harbour of the power station is navigable by vessels with coal cargoes as heavy as 120,000 tonnes. The plant is supplied with a new, highly automated, mechanised coal handling system. Coal is unloaded to a coal yard with an unloading capacity of 1,000 tonnes per hour. From there coal is loaded on belt conveyors leading to the silos in the boiler plant via the crushing stations.

The limestone is mainly extracted in Estonia and Gotland and milled in Finland.

4.1.3. Downstream fuel-cycle operations

Ash formed at the power station totals about 150,000 tonnes annually. The ash is delivered as raw material to the cement industry or is utilised in landfills. The amount of gypsum, the end product of the desulphurization plant, has grown by about 60,000 tonnes per year. The gypsum is sold as raw material to a gypsum board factory. Ash and gypsum are transported by lorries.

The cooling water, 14.5 m^3/s, is returned to the sea. The cooling water is mechanically cleaned sea water and the temperature increase of cooling water in the condenser is about 10°C.

4.1.4. Site description

The Meri-Pori power plant is located on the island Tahkoluoto within the city of Pori on the west coast of Finland. The geographical coordinates of the plant are: latitude 61.63°, longitude 21.41°. The population density is 152 inhabitants per square kilometre in Pori but lower in its surroundings.

4.2. Overview of burdens

The major burdens of the coal fuel cycle are the atmospheric emissions of pollutants from the mining and power generation stage, liquid effluents and solid wastes from mining and power generation, and occupational accidents from the mining stage. The burdens of the coal fuel cycle are described stage by stage in Table VIII.2 of Appendix VIII. In this chapter an overview of the different burden types is given.

4.2.1. Air emissions

The specific emissions of the power generation stage at the Meri-Pori power plant used in the calculations are given in Table 4.2. The NO_X and TSP numbers are close to the official emission standards (Meri-Pori power plant, 1995) and give an upper estimate of the actual emissions. However, according to the air protection declaration of the plant for the year 1994, the actual specific emissions of TSP appeared to be essentially smaller: only 0.013 g/Nm^3 compared to the value 0.05 g/Nm^3 used in this study. The emissions of CO_2 and CH_4 were estimated from the general specific emissions of coal: the combustion of coal generates greenhouse gases at a rate of 93 g CO_2/MJ_{fuel} and 0.005 g CH_4/MJ_{fuel}.

Table 4.2 Air emissions of the power generation stage and their percentage of the whole coal fuel cycle emissions.

	Emissions t /a	Specific emissions g/kWh_e	Specific emissions g/Nm^3	% of the whole coal fuel cycle
SO_2	2,400	0.67	0.226	81%
NO_X	1,900	0.53	0.180	84%
TSP	540*	0.15*	0.050*	88%
CO_2	2,800,000	770	260	97%
CH_4	150	0.04	0.014	1.4%
N_2O	60	0.017	0.006	100%(?)

*The actual emissions appear to be only about 1/4 of these numbers.

As can be seen the main part of the emissions to the air are caused by the power generation stage except in the case methane emissions.

Fugitive dust, not shown in Table 4.2, is also a burden addressed to the other fuel cycle stages. The dust emissions were estimated to be as much as 6,000 t/a, of which 72% was allocated to the coal storage (at the power plant site), 22% to the transportation stage and 6% to the fuel production. However, these appear to be very uncertain estimates, and the impact of fugitive dust was *not* assessed in this study.

Table 4.3 Air emissions of the other stages of the coal fuel cycle.

	Emissions t /a	Specific emissions g/kWh_e
SO_2	580	0.16
NO_x	370	0.10
TSP	75	0.02
Dust	6300	1.73
CO_2	92000	25
CH_4	10000	2.8
N_2O	nq	nq
CO	40	0.012
HC	30	0.008

Outside the combustion stage, the major part of the SO_2 emissions originate from fuel production whereas the different transportation stages are responsible for most of the NO_x emissions.

Information on environmental impacts of coal mining in Poland is based on the IEA reports and on discussions with specialists working in Katowicki holding Wlglowy S.A. in Poland (Ekono, 1996). The gas emissions from coal mining were estimated by using the energy consumption figures of coal mining and the emission limits for energy generation plants in Poland. Methane seepage is a serious problem in underground mines in both Poland and Russia. Methane can ignite if it is not extracted through a ventilation system. Over two-thirds of the underground mines in these countries have gas emission rates of over 10 m^3/t coal. (Doyle, 1989) The emissions of greenhouse gases from Polish coal mines are estimated at 0.33—0.35 g CH_4/MJ_{fuel} and 2 g CO_2/MJ_{fuel}.

Other atmospheric emissions from coal mining are estimated at 20 mg SO_2/MJ_{fuel}, 3 mg NO_2/MJ_{fuel} and 16 mg TSP/MJ_{fuel} using the energy consumption figures of mining and the emission limits for Polish coal-fired power plants.

The atmospheric emissions from ship and rail transportation of coal are estimated at 4 mg SO_2/MJ_{fuel}, 25 mg NO_2/MJfuel and 140 mg TSP/MJ_{fuel}. Serious environmental impacts of transportation are probably caused by dusting, but these were not considered further.

Emissions (and their impacts) from power plant construction and demolition were not estimated in the Finnish national implementation. According to the previous ExternE results (European Commission, 1995c) these impacts are small compared to the power generation stage during the lifetime of the plant.

4.2.2. Waste water and solid waste

The presence of water and gas in many mines creates further difficulties and hazards. Many Polish mines have to pump substantial quantities of water to the surface for desalination and disposal (Doyle, 1989). Large quantities of mine water with high salinity are drained directly into rivers. The rivers of Upper Silesia are saturated with salt and their water cannot be used even in agriculture. Desalination plants to treat wastewater from coal mining will be built in the near future.

Solid mine waste piles were a major environmental problem in Poland. Currently, the dumping of waste back into the mines is cheaper than storage in overground piles because of taxes.

Rock may fall in the abandoned mine galleries, which can also cause subsidence.

The solid waste and cooling water from the power generation stage appear to be only a minor problem.

4.2.3. Direct public and occupational health effects

In addition to the aerial emissions and other waste flows with indirectly public health impacts operation of the coal fuel cycle causes direct burdens, for example in the form of mine and traffic accidents. The rate of mine accidents in Poland was obtained from the Statistics of ILO (1995), see Table 4.4. About 75 per cent of employees working in mines are working underground. Black lung is the most common occupational disease among coal workers.

The accidents during the transportation of fuel (by rail and ship) were not considered in this study mainly because their incidence (per t of load) is much lower than that of truck accidents. The truck accidents due to limestone, gypsum and ash transport in Finland were assessed and are presented in Table 4.4.

Table 4.4 Accidents of the coal cycle per TWh of electricity produced.

	Fatal accidents	Major injuries
1. Coal mining	0.19	6.3
2. Coal transport	nd	nd
4. Limestone extraction	nd	nd
5. and 8. Transport (pa, w)	0.008	0.03
6. Power generation	nd	nd
10-11. Construction and dismantling	nd	nd

nd : not determined

4.2.4. Noise

The noise level does not exceed 45dB(A) at a distance of 100 m from the power plant and cannot be considered as causing any negative impacts or damages. Noise might be a serious problem in the mining and transport stages of the fuel cycle but no information on these noise levels was gathered from literature.

4.3. Selection of priority impacts

A list of major impacts of the coal fuel cycle is presented in Table 4.5.

According to previous studies of the ExternE Project (European Commission, 1995abcdef) and the experiences of the National Implementation projects the most important impacts of the coal fuel cycle are those caused by atmospheric emissions, especially from the power generation stage. Liquid effluents from mining in Poland appear to have serious local environmental impacts but their quantification could not be performed in this study. Applying the ExternE methodology the human health impacts seem to dominate over those directed to other recipients. Occupational health impacts in Polish mines are apparently the most important ones of the whole coal fuel cycle.

Table 4.5 Principal impacts of the coal fuel cycle.

Impacts	C & L mining	Transport of PA&W	Power generation	Waste disposal	Construction/ demolition
Global warming	x	x	x	x	x
Public health		x	x	x	x
Occupational health	x	x	x		x
Crops	x	x	x		x
Forests		x	x		x
Ecosystems	x	x	x	x	x
Materials		x	x		x
Noise	x	x	x		x
Road traffic		x		x	x
Visual impact	x		x		x

The impacts identified as primary for the Finnish coal fuel cycle are:

- global warming due to release of greenhouse gases during coal mining, transport and combustion
- effects of atmospheric pollution on human health
- injuries and occupational health risks to workers and the general public
- effects of air pollution and acidic deposition on the built-up environment
- effects of air pollution on crops
- effects of air pollution and acidic deposition on natural ecosystems
- impacts of coal mining on ground and surface water quality
- impacts of coal mining on soil

Because of the lack of quantitative information on mining and its environmental impacts in Poland the last two impact groups could not be assessed in this study.

4.4. Quantification of impacts and damages

The model EcoSense 2.0 was the tool for calculating the dispersion of the primary pollutants TSP, SO_2 and NO_x — and the consequent impacts and damages — of power generation.

Because the computational grid of the model does not cover areas east of Finland, all the air emission impacts, e.g. in Russia are missing in the basic model results. This would lead to an underestimation of impacts and damages especially in the Finnish national implementation. Consequently, there was a need to extend the model data base to include the Russian data.

It was decided to take into account at least the Russian health impacts, because they appeared to be so important. The population of north-west Russia, i.e. the city of St. Petersburg and the regions of Leningrad, Karelia and Murmansk (totally about 8.4 million people) was added to the easternmost gridcells of the model. The health damage numbers in the Tables of this chapter and in Table VIII.3 in Appendix VIII include this population.

EcoSense model calculations were also applied to other fuel cycle stages. As an approximation it was assumed that their emissions (e.g. from transport) take place through the power plant stack. The dispersions of the primary pollutants (TSP, SO_2, NO_x) were then calculated with EcoSense.

Ozone formation is not included in the EcoSense version 2.0 applied in the study. Ozone formation was assumed to be directly proportional to the NO_x emissions. The average damage of ozone in the whole European scale was estimated to be 1500 ECU/ t NO_x (Rabl and Eyre, 1997), of which the damage due to mortality was about 415 ECU/ t NO_x, morbidity 735 ECU/ t NO_x and crops 350 ECU/ t NO_x.

The assessment of global warming damage is described in Appendix V.

In the following the impacts and their damage estimates are discussed by impact category. The summarised estimates for coal cycle is given in the Tables and Figures at the end this Chapter. More detailed results are presented in Table VIII.3 of Appendix VIII. The estimated damages of different impact categories (human health including accidents, crops, materials, global warming) are illustrated in Figures 4.1 and 4.2.

4.4.1. Public and occupational health

Mortality due to airborne emissions was valued by using the value of a life year (VOLY) and in addition on the basis of the value of statistical life (VSL), where VSL was equal to 3.1 million ECU. In fatal accidents only the VSL principle was applied. The theoretical basis and details of the valuation of health effect are described in Appendix II.

Public health damages are a consequence of the impacts of air emissions: the emissions of power generation and all other stages which increase mortality (VOLY) and morbidity. In addition, health impacts of road accidents caused by the transport of limestone, gypsum and ash in Finland were included in these figures but train and ship accidents were excluded. Only the VSL principle was applied to the accidents.

Occupational health damages consist only of the accidents and occupational diseases in Polish mines. The damages of occupational accidents and diseases in Finnish power plants were assumed to be a negligible factor.

The damage value applied to public and occupational major accidents in this study was 95 000 ECU, as presented in Table II.12 in Appendix II.

The health damages of power generation are given in e following Tables and Figures.

Table 4.6 and Table 4.7 and results from the whole coal cycle in Table 4. and Table 4. Power generation is responsible for the major part of the damages of the fuel cycle: its share appears to be 73%, using the VSL approach (in valuation of mortality) and 59%, using the VOLY approach, of the damages caused by airborne pollutants. The VSL approach was always applied to the accidents, so that damage comparisons of health impacts within the whole fuel cycle cannot be made using VOLY only.

An important damage factor of the Finnish coal cycle are the mining accidents in Poland. These comprise about 12% of all the health damages using the VSL approach and 28% if the VOLY approach is applied to the illnesses linked to air pollution. The mining accidents represent a threat to a smaller number of people. An open question is whether this damage is internalised in the costs of coal mining or not.

4.4.2. Global warming

Global warming is the most important damage factor of the Finnish coal cycle when applying the ExternE methodology, but the knowledge behind all global warming estimates is rather poor at present. It is actually one of the most uncertain components of the ExternE methodology. The four damage estimate figures per tonne of CO_2 emission which were used in the ExternE National Implementation are given in Table V.4 of Appendix V. The impacts of CH_4 and N_2O were calculated from the CO_2 damage estimate by multiplying it with their $GWP_{100\ a}$ coefficients. In addition, after finishing the calculations and NI projects (end 1997) it became clear that there might be some methodological confusion behind all the GW damage numbers presented in Appendix V.

The damage estimates for the Finnish coal cycle are shown in Figure 4.2, where a comparison to health damages is also given. The power generation stage causes 90% of the greenhouse impacts of the whole chain (when the using the GWP_{100} coefficients for the greenhouse gases), illustrated in Figure 4.3. Methane emissions from Polish mines are also a significant factor.

4.4.3. Other damages evaluated

The impacts on crops via ozone appear to be the most important of the quantified damages.

No monetary measures could yet be developed in the ExternE National Implementation Study for ecosystem damages. Only the impacts on ecosystems were quantified in the form of increase in land area where the critical load of acidity was exceeded. The impacts of power generation are given in e following Tables and Figures.

Table 4.6 and results for the whole fuel cycle in Table VIII.3 of Appendix VIII. It should be remembered that all the land areas east of Finland are here excluded. The ecosystem impacts (not valued in monetary terms) caused by the Meri-Pori power station are briefly discussed in chapter 3.4.

The valuation of the material damages is based on the database of EcoSense, which includes the surface areas of different kinds of building materials and their monetary values. The building materials are exposed by the acid deposit, which causes the damages. Road damages caused by transport were not assessed in the Finnish implementation.

4.4.4. Summary figures of power generation and the whole coal cycle

The impact and damage estimates of the power generation stage and whole fuel cycle are illustrated by the following Tables and Figures.

Table 4.6 Sub-total damages of power generation by impact categories (north-west Russian population included).

Impact	Burden	Impacts		Damages		
		Unit	per TWh	mECU/kWh	ECU/t	σ_g
Human health						
Chronic VOLY	TSP	years	2.4	0.20	1378	B
	Nitrates	years	7.3	0.62	1160	B
	Sulfates	years	9.8	0.83	1240	B
Acute VOLY	SO_2	years	0.16	0.03	38	B
	Ozone			0.22	415	B
Morbidity	Nitrates	cases	457	0.08	150	A
	Ozone			0.39	735	B
	SO_2	cases	nd			
	Sulfates	cases	623	0.10	155	A
	TSP	cases	151	0.026	178	B
Crops	SO_2	dt yield loss	444	0.003	5	A
	Ozone			0.19	350	B
	N dep.	kg fertiliser added	-3308	-0.00006	-0.1	A
	Ac. dep.	kg lime added	27670	0.0005	na	A
Ecosystems	N	km^2 exceed. area	162	na	na	
	SO_2	km^2 exceed. area	42	na	na	
Materials	SO_2	m^2 maint. area	2435	0.032	48	B
Occupational accidents			nd			
Global warming	CO_2					C
Conservative 95% confidence interval			low	2.9	3.8	
			high	107	139	
Illustrative restricted range			low	14	18	
			high	35	46	

na = not applicable, nd = not determined,
σ_g = standard deviation confidence band (APP VII)
VOLY = ‘value of life year’ approach

*Table 4.7 Sub-total health damages of power generation (no accidents considered, north-west Russian population included)***.**

		mECU/kWh
VOLY ***(VSL)***	low	nd
	mid	**2.5*** ***(6.9**)***
	upper	nd

*VOLY= mortality impacts of airborne emissions based on 'value of life year' approach
***VSL* = mortality impacts of airborne emissions based on 'value of statistical life' approach
nd = not determined

Table 4.8 Damages of the whole coal fuel cycle (north-west Russian population included).

	mECU/kWh	σ_g
POWER GENERATION		
Public health		
Mortality*- VOLY *(VSL)*	1.9 *(6.3)*	B
of which TSP	*0.20 (0.7)*	
SO_2	*0.85 (3.1)*	
NO_x	*0.62 (2.3)*	
NO_x (via ozone)	*0.2*	
Morbidity	0.6	
of which TSP, SO_2, NO_x	*0.2*	A
NO_x (via ozone)	*0.4*	B
Accidents	nq	A
Occupational health	nq	A
Crops	0.2	B
of which SO_2	*0.003*	
NO_x (via ozone)	*0.2*	
Ecosystems	iq	B
Materials	0.03	B
Monuments	nq	
Noise	ng	
Visual impacts	nq	
Global warming		C
low	2.9	
mid 3%	14	
mid 1%	36	
high	108	
OTHER FUEL CYCLE STAGES		
Public health		
*Outside EU***	*0.5 (1.4)*	B
*Inside EU****	*0.03*	B

Table 4.8 Continued

Occupational health		A
*Outside EU*** **	*1.2*	
Inside EU	*nq*	
Ecological effects	nq	B
Road damages	nq	A
Global warming		C
low	0.32	
mid 3%	1.5	
mid 1%	3.9	
high	12	

*VOLY= mortality impacts based on 'years of life lost' approach, VSL= impacts evaluated based on 'value of statistical life' approach. VSL only applied to fatal accidents. σ_g = standard deviation confidence band (see Appendix VII). **air emissions of fuel production and transport; ***road accidents in Finland due to transport of ash, gypsum and limestone. ****mining accidents in Poland; ng: negligible; nq: not quantified; iq: only impact quantified.

Table 4.9 Total health damages of the coal fuel cycle **(***north-west Russian population included)***.**

		mECU/kWh
VOLY *(VSL)*	low	nd
	mid	**4.2* *(9.6**)***
	upper	nd

*VOLY= mortality impacts of airborne emissions based on 'value of life year' approach, mortal accidents based on 'value of statistical life' (VSL) approach, ***VSL*= all mortal impacts evaluated based on 'value of statistical life' approach , nd = not determined

*Table 4.10 Total quantified damages by pollutant (north-west Russian population included, no ecosystems)***.**

	ECU / t of pollutant
SO_2 *- VOLY *(VSL)*	1486 *(4841)*
NO_x *- VOLY *(VSL)*	1310 *(4416)*
TSP *- VOLY *(VSL)*	1555 *(5242)*
NO_x (via ozone)	1500
CO_2	3.8—139

*VOLY= mortality impacts of airborne emissions based on 'value of life year' approach,
VSL = mortality impacts of airborne emissions based on 'value of statistical life' approach

*Table 4.11 Mortal damages by pollutant (north-west Russian population included)***.**

	ECU / t of pollutant
SO_2 *- VOLY *(VSL)*	1278 *(4633)*
NO_x *- VOLY *(VSL)*	1160 *(4267)*
TSP *- VOLY *(VSL)*	1378 *(5064)*
NO_x (via ozone)	415
CO_2	na

na = not applicable

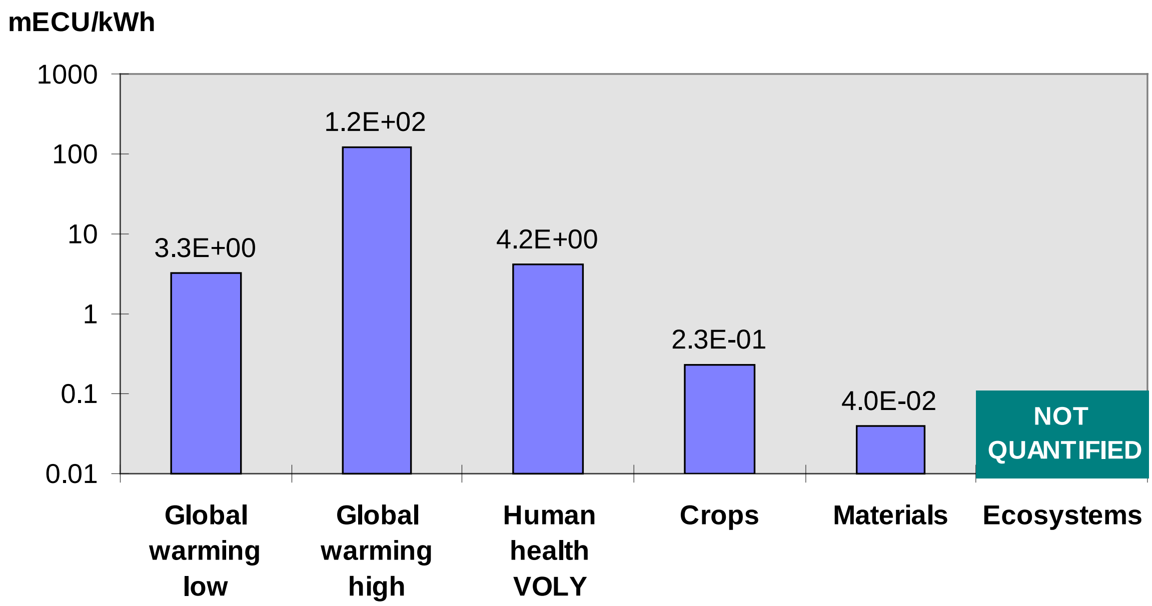


Figure 4.1 Total damages of coal fuel cycle by impact category.

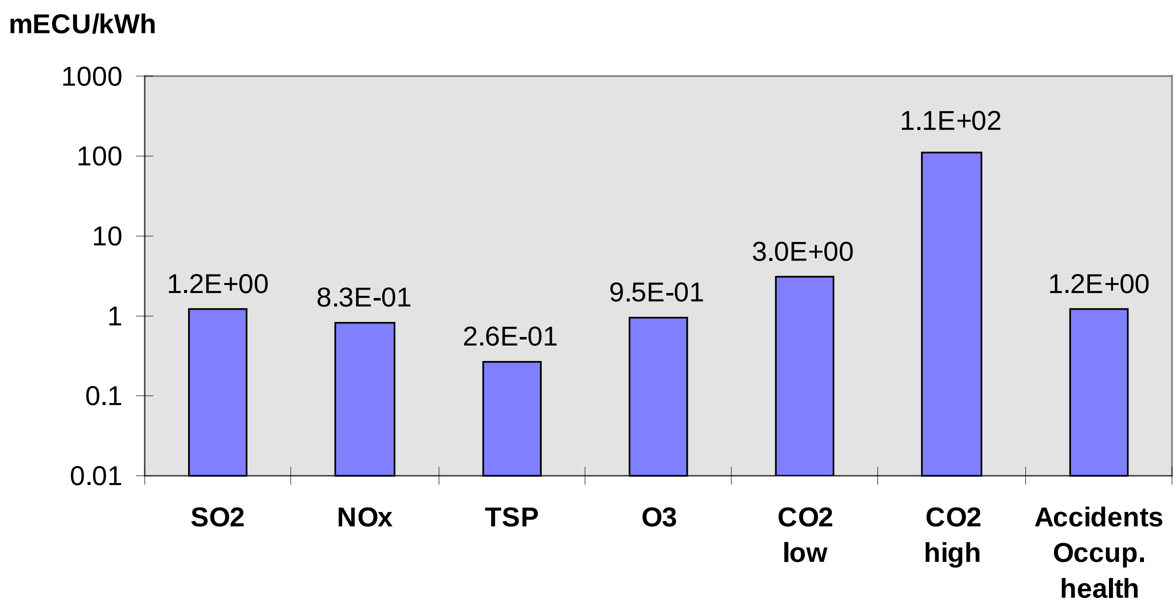


Figure 4.2 Total damages of coal fuel cycle by burden category.

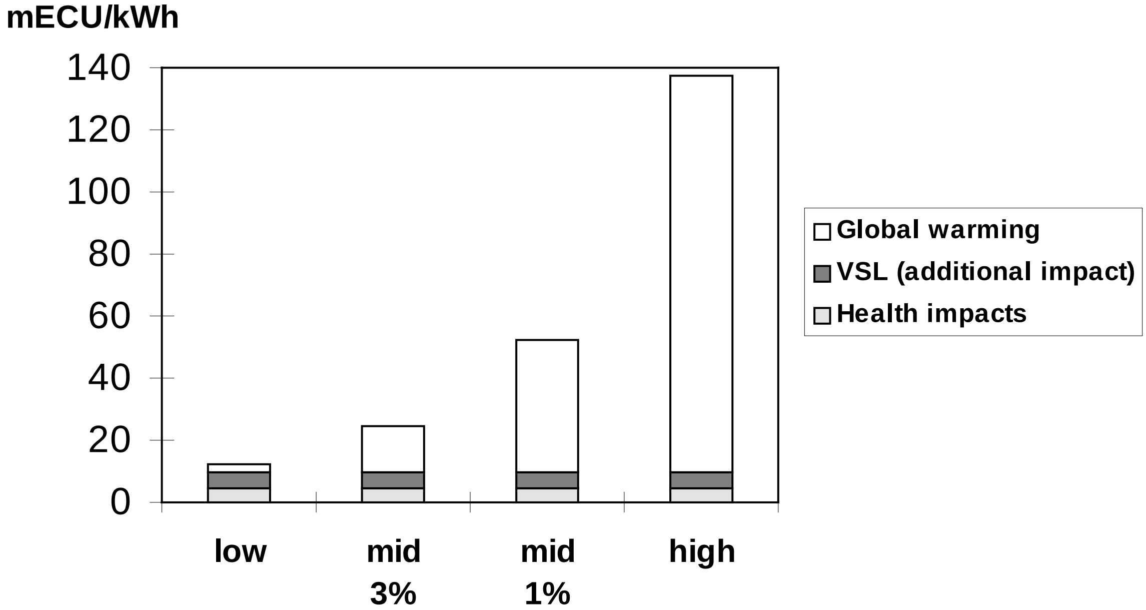


Figure 4.3 Estimates of global warming damages vs. health damages per kWh of electricity.

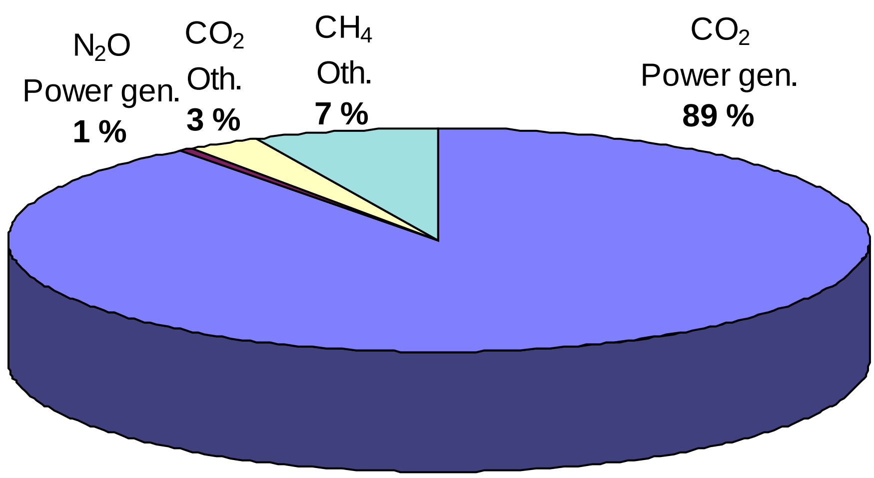


Figure 4.4 Relative impacts of the global warming components ($GWP_{100\ a}(CH_4) = 21$, $GWP_{100\ a}(N_2O) = 310$).

4.5. Interpretation of the results and sensitivity analysis

The damage estimates in the ExternE methodology are dominated by health impacts and global warming as demonstrated in Figures 4.1 and 4.2. The theoretical work in the ExternE Project also paid particular attention to health impacts. The impacts on ecosystems seem to be very difficult to assess, especially the long-term impacts. An even more difficult task is to give any monetary valuation to this kind of damages. The evaluation of material damages could be realistic if the database of EcoSense included reliable data on building materials in Europe, but this remained an open question to the Finnish team (as well as the data concerning natural ecosystems).

With the exception of the mining stage the accidental damages seem to be negligible in the Finnish coal cycle, compared to the other health impacts. The transportation of fuel by rail and ship means lower accident rates. The truck transports (with higher accident rates) represent only a negligible part of the total transports. However, the exact damage of fuel transport was not assessed. The occupational accidents in Finnish power plants and limestone mining were also considered as minor factors (not valued) in the whole fuel cycle.

The EcoSense model allows a calculation of the TSP, SO_2 and NO_x impacts and damages in Western Europe by taking into account the dispersion and chemical reactions in the air. As mentioned earlier the TSP emissions appear to be overestimated in this study leading to similar overestimates in their impacts and damages. The calculation of ozone damage is at present much coarser and the results an order of magnitude more uncertain. The same damage estimate ECU/t (proportional to the emitted amount of NO_x) was applied all over Europe, which may cause an overestimate of damages caused by Finnish power plants. Therefore obviously the ozone damages to crops seem to totally dominate over the SO_2 damages in the calculations.

The estimates of global warming are an order of magnitude more uncertain. Although the emissions of greenhouse gases and their global warming potentials (GWPs) are well known, the knowledge of the true impacts and damages of GW is still poor.

The approximation that the pollutant from other stages also emits through the stack probably underestimates slightly the impacts of the emissions, which in reality occur in more densely populated areas and closer to the ground.

One problem concerning the Finnish national implementation were the limits of the computational grid in EcoSense (Finland being situated in one edge of the grid), which was assumed to underestimate the real damages. As the winds on the average come from south west, the emission impacts outside the Finnish borders mainly occur to the north east of the country. Therefore, it seemed to be a fair approximation to place the Russian population in the crowded provinces east from Finland to the most eastern cells in the model, although this slightly exaggerates the damage results by bringing the additional population closer to the plant than in reality. Although the sparsely populated land areas in Russia more further to the east were not included, it is obvious that the population included in the model extension is the major recipient of the health impacts of the Meri-Pori power plant.

The sensitivity of the damage estimates to this additional population of 8.4 million people was also investigated. After adding the Russian population the damages of SO_2 and TSP were increased by 17% and the damages of NO_x by 31%. This indicates clearly how the impacts of NO_x emissions extend further than those of the other two pollutant components.

The specific emissions of the Meri-Pori plant used in the EcoSense model may overestimate the real emissions. According to the air protection declaration of 1994, for example, the true TSP emissions might be only 1/5 of the emission standards, which were used in EcoSense runs. This means that the corresponding damages would also be about 1/5 of those given in Table 4. and Table 4.10. Correspondingly, the damages of NO_x emissions could be about 30% lower than those given in the above Tables.

5. Peat fuel cycle

5.1. Definition of the fuel cycle

A detailed definition of the Finnish peat fuel cycle is given in Appendix IX, where the burdens, impact and damage figures of the cycle are also listed.

The principal reference year is 1995. The meteorological data for the ISC model of the EcoSense are from 1993.

5.1.1. Power generation stage

The Rauhalahti power plant is located in the Jyväskylä area and is owned by Jyväskylän Energiantuotanto Oy. The power plant is a cogeneration plant producing electricity, heat and steam. The plant has a bubbling fluidised bed boiler. The main fuel is milled peat, but the new combustion technique enables the utilisation of wood fuels such as sawing waste, chips and bark alongside peat. Crushed coal and oil can also be burnt in the boiler. The thermal power of the boiler is 295 megawatts. The power plant produces 87 MW_e (83 MW_e sent out) electricity, 125 MW_h district heat and 45 MW_s process steam. The power plant was commissioned in 1986 to burn pulverised peat, but was converted to a *bubbling fluidised bed* boiler in 1993. The efficiency of the plant is about 85% (Rauhalahti Power Plant, 1994).

The plant uses over one million cubic metres of milled peat per year, meaning about 17 000 hectares during its 30-year lifetime. The peat is transported by trucks from peatlands in the vicinity of the plant, with an average transportation distance of 80 km. The plant consumes about 60 truck loads of peat per day. In 1995 its overall fuel consumption was about 84% milled peat, 13% wood, 2% oil and 1% coal. The operating time in 1995 was 8328 hours (Mälkki and Frilander, 1997).

The flue gases of the power plant go through the electrostatic precipitator, which separates over 99% from the ash. The fluidised bed boiler reduces the nitrogen oxide emissions formed during combustion by over one third compared with the earlier pulverised boiler. The wood fuels form very little sulphur emissions and the amount of fossil carbon emissions in the air are smaller when wood is used.

In the power plant there is also a small oil boiler for producing process steam when the main boiler is occasionally non-operational (not considered in ExternE).

5.1.2. Upstream fuel cycle stages

The peat fuel cycle begins from the natural peatland. The upstream fuel cycle includes ditching, preparation, profilation, production, stockpiling and transportation.

Ditching removes free water from the natural peatland. The ditching of bogs is carried out mainly with a 20 m strip width using open-strip ditches. The groundwater level and the moisture content of the field are lowered. The peat is dried on the surface of the field. The draining period lasts about 3—6 years. With the present amount of open-strip ditches, the ditches take up about 10% of the production area (Leinonen & al, 1993). The preparation phase for peat production begins when the moisture content of the peatland is lowered from 92% to 82%. In this phase the environmental actions are also completed and the surface of peatland is profiled.

Table 5.1 Fuel characteristics of milled peat, coal, oil and wood waste (Mälkki and Frilander, 1997).

1995	Milled peat	Coal	Oil	Wood waste
total amount (t)	519768	1861	418	93584
transportation mode	truck	rail	truck	truck
transportation distance in Finland (km)	50—100	269	326	0—200
Caloric value (MJ/kg)	10.1	23.8	40.7	8.4
moisture (%)	46.3	8—9		40—55
sulphur content (% of dry matter)	0.23	0.87	0.5	0.05
ash content (% of dry matter)	5.2	12—14		0.4—2.8

Table 5.2 Electricity, heat and steam produced in the Rauhalahti power plant in 1995 (Mälkki and Frilander, 1997).

Fuel need (GWh/a)	1669
Thermal power (MW)	295
Boiler output (MW)	266.8
District heat (MW)	125
Process steam (MW)	45
Electricity sent out (MW)	83
Electricity for own use (MW)	4
Operating hours per year (h)	8328
Full load hours per year (h)	5655

Production of milled peat is nowadays performed mainly by the Tehoturve method. The method was developed in 1992 to improve the production efficiency of milled peat on the basis of the traditional Haku method. The processed peat from the fields is stored in stockpiles. During the storage time the stockpiles may become moist. Furthermore the peat in the stockpiles may be self-heated and may ignite. The peat quality may decrease and energy losses are also inevitable. The milled peat is normally transported by trucks to the power plant. The average transportation distance is 80 km.

5.1.3. Downstream fuel cycle stages

The downstream fuel cycle includes restoration of the peatland and waste management from energy generation. Utilised peatland can be restored by different means: afforestation, agricultural usage, energy plants, recreation areas, wetland areas for birds and paludification. Afforestation is currently most common, but rewetting of peatland is also becoming more popular. The rewetting mode is used in this study.

The energy generation stage produces wastes as slag and ash. The wastes are used locally mainly for earth filling. The transportation of waste is not considered in this study, because the distances are very short.

5.1.4. Site description

The Rauhalahti power plant is located in Central Finland, near to the city of Jyväskylä. The geographical coordinates of the plant are: latitude 62.23^o, longitude 25.81^o.

5.2. Overview of burdens

The calculated emissions are net emissions. The emissions of natural peatland are subtracted from the calculated emissions of the other stages of the peat fuel cycle. Natural peatlands are net sinks for carbon dioxide but sources of methane and nitrous oxide emissions. The amounts of the emissions depend on the season. During the winter season emissions are about 10% of their levels in summer and also during daytime they are 50—100% higher than at night. Photosynthesis ceases at night and during the winter. However, the greatest amounts of SO_2, NO_x and CO_2 emissions originate mainly from the energy conversion phase.

The values of the greenhouse gases are taken from the most recent studies and are consistent with the SILMU reports (SILMU = The Finnish Research Programme on Climate Change) (Kuusisto et al., 1996). The following tables (Table 5.3**,** Table 5.4 and Table 5.5) present the values for greenhouse gases.

The greatest share of the CO_2 emissions comes from the energy conversion process of the power plant. According to the results, the contribution of energy conversion to the total CO_2 emissions is between 95 and 98%. The shares of ditching and preparation and peat transportation are below 1% and the share of peat production is about 5%. Stockpiling accounts for about 5% of the total CO_2 amount. The CO_2 share of the restoration stage depends on the selected mode of restoration. Afforestation is shown to be only slightly more favourable than rewetting, if the restoration period is considered to be 100 years. The sink effect of the restoration phase is about 6—10% of the total CO_2 emissions. The air emissions of the peat fuel cycle are summarised in Tables 5.6 and 5.7. Actually the TSP emissions may be larger than given in Table 5.6, e.g. in 1997 nearly double (Järvinen 1999). The quantified burdens are given in Table IX.2 in Appendix IX.

Table 5.3 CH_4 emissions.

CH_4 (g CH_4 /m_2 /a)	PHASE OF PROD.	REFERENCE	REMARKS
7	Before ditching	Hillebrand 1993	Depends strongly on original mire type.
0.3	After ditching	Kuusisto et al., 1996	Depends on original mire type.
4	Rewetting	Hillebrand ja Vihersaari 1993	

Table 5.4 CO_2 emissions.

CO_2 (g C /m^2 /a)	PHASE OF PROD.	REFERENCE	REMARKS
-20	Before ditching	Hillebrand 1993	Depends strongly on original mire type.
250	After ditching, production	Kuusisto et al., 1996	High value, depends strongly on original mire type.
-64	Rewetting	Hillebrand ja Vihersaari 1993, Roderfeld et al 1994	
-20.5	Mineral subsoil	Turunen et al. 1996	

Table 5.5 N_2O emissions

N_2O (g N_2O /m^2 /a)	PHASE OF PROD.	REFERENCE	REMARKS
0.015	Before ditching	Hillebrand 1993	Depends on original mire type.
0	After ditching	Hillebrand 1993	
0	Rewetting	Hillebrand ja Wihersaari 1993	

Table 5.6 Air emissions of the power generation stage and their percentage of the whole peat fuel cycle emissions.

	Total emissions t /a	Specific emissions (exergy allocation) g/kWh_e	Specific emissions g/Nm^3	% of the whole peat fuel cycle
SO_2	1,300	1.9	0.57	99%
NO_X	760	1.1	0.33	92%
TSP	90*	0.12*	0.037*	92%
CO_2	630,000	900	270	96%

*****The actual TSP emissions may be somewhat greater.

Table 5.7 Air emissions of the other stages of the peat fuel cycle.

	Emissions t /a	Specific emissions g/kWh$_e$
SO_2	20	0.03
NO_x	60	0.09
TSP	10	0.01
CO_2	30,000	40
CH_4	-600	-0.8
N_2O	30	0.04

5.2.1. Water emissions

Water emissions from peatlands to surrounding areas are caused by ditching, preparation and peat production. Emissions have been decreased in recent years by the new water protection methods. Solids outwash is prevented by ditch retainers, sedimentation, filtering, soil infiltration or by surface runoff. All new peatland areas are provided with settling ponds constructed in accordance with regulations. Furthermore, old peat harvesting areas are refined and updated by the water protection measures. The settling pond may reduce the amount of solids by 30—40%, but it does not affect the amount of dissolved nutrients. In experiments, drainage waters have been purified with the same chemicals as those used to treat drinking water. The treatment resulted in even better water quality than the runoff from a natural peatland. However, the method is expensive and thus applicable only to water treatment at large production sites (Vasander (ed.), 1996).

The water emissions (Table 5.8) are calculated by Vapo Oy on the basis of the control monitoring reports. Reports are available for the years 1994 and 1995. The emissions also depend on the various measures effected for environmental protection. The water emissions of the restoration phase are not available at present.

Table 5.8 Average water emissions according to the control monitoring reports in Finland (Vapo Oy, 1997).

Vapo OY - FINLAND	SOLIDS (kg/ha/a)	TOT. P (kg/ha/a)	TOT. N (kg/ha/a)	COD Mn (kg/ha/a)	NH4-N (kg/ha/a)
Average value (1994—1995)					
Production	41.08	0.21	5.19	77.91	3.36
Ditching and preparation	83.82	0.39	7.78	143.09	2.84
Resting state, forest drained	11.65	0.08	1.62	64.57	0.62
Background, natural state	4.95	0.07	1.33	36.29	0.05

5.3. Selection of priority impacts

A list of principal impacts of the peat fuel cycle is presented in Table 5.9.

Table 5.9 Principal impacts of the peat fuel cycle.

Impacts	Peatland ditching, preparation	Peat production, stockpiling	Transport	Power generation	Peatland restoration	Waste disposal	Construction / demolition
Global warming	x	x	x	x		x	x
Public health			x	x		x	x
Occupational health	x	x	x	x	x		x
Crops			x	x			x
Forests			x	x			x
Fisheries	x	x			x		
Ecosystems	x	x	x	x	x	x	x
Siltation	x	x			x		
Materials			x	x			x
Fires		x					
Noise	x	x	x	x			x
Land use	x	x			x	x	x
Road traffic			x			x	x
Visual impact	x	x		x			x

The most important impacts of the peat fuel cycle are those caused by atmospheric emissions, especially from the power generation stage. Liquid effluents from peatland ditching and peat production have environmental impacts such as eutrophication of the neighbouring water systems. Applying the ExternE methodology the human health impacts appear to dominate those directed to other recipients. Occupational health impacts of peat production are not considered in this study.

The impacts identified as primary for the Finnish peat fuel cycle are:

- global warming due to release of greenhouse gases during peatland ditching, preparation, peat production, stockpiling, transportation and combustion
- effects of atmospheric pollution on human health
- injuries to the general public from the transportation of peat
- effects of air pollution and acidic deposition on the built-up environment
- effects of air pollution on crops
- effects of air pollution and acidic deposition on natural ecosystems
- effects of water pollution on natural ecosystems and fisheries
- effects of land use on natural ecosystems and recreation areas
- effects of siltation on natural ecosystems and water systems
- impacts of peatland ditching and peat production on ground and surface water quality
- impacts of peat production on soil

Because of a lack of quantitative information on impacts in Finland, the last five impact groups could not be assessed in this study.

5.4. Quantification of impacts and damages

The main tool for calculating the dispersion and air chemical reactions of the basic pollutants TSP, SO_2 and NO_x — and the consequent impacts and damages — was the model EcoSense 2.0. Because the computational grid of the model does not cover areas east of Finland, all the air emission impacts e.g. in Russia are missing in the basic model results. This leads to an underestimation of impacts and damages especially in the Finnish national implementation. Consequently, there would be a need to extend the model data base with Russian data.

In the Externe methodology the impacts and damages are allocated to electricity and heat using the *exergy* principle, which appears to mistreat the electricity generation in CHP plants in proportion to condensing plants. As a result of this most of the impacts/damages were assigned to electricity in the Rauhalahti case as can be seen in Table 5.10. In Finnish energy statistics the allocation is different and based on *energy* content.

Table 5.10 Rauhalahti peat fired power plant: allocation of impacts/damages (exergy principle applied in this ExternE NI study).

	Production per year GWh	Allocation principle: <u>*Exergy*</u> % of impacts/ damages allocated	<u>*Energy*</u> % of impacts/ damages allocated
Electricity sent out	469	**67.6%**	32.8%
District heat	707	**21.8%**	49.4%
Process steam	254	**10.7%**	17.8%

*Table 5.11 Total health damages of the peat fuel cycle north-west Russian population excluded)***.**

		mECU/kWh
VOLY *(VSL)*	low	nq
	mid	4.9 *(12.1)*
	upper	nq

The air emissions (TSP, SO_2, NO_x) of the other stages were also calculated with EcoSense by assuming that all these emissions (e.g. from transport) were released from the power plant stack.

Ozone formation is not included within the EcoSense version 2.0 applied in the study. Ozone formation was assumed to be directly proportional to the NO_x emissions. The average damage of ozone in the whole European scale was estimated to be approximately 1500 ECU/ t NO_x (Rabl and Eyre, 1997), of which 415 ECU/ t NO_x due to mortality, 735 ECU/ t NO_x to morbidity and 350 ECU/ t NO_x due to crop damage.

Global warming is the most important damage factor of the Finnish peat cycle when applying the ExternE methodology. The recommended estimates (by the Core Project) for the ExternE National Implementation studies are described in Appendix V.

In the following the impacts and their damage estimates are discussed by impact category. The summarised estimates for peat cycles are given in Table 5.6, Table 5.7 and Table 5.8. More detailed results are presented in Table IX.3 of Appendix IX.

5.4.1. Public and occupational health

The damage due to mortality was in principle valued by using the value of years of life lost (VOLY) but the valuation was also performed on the basis of the value of statistical life (VSL), where VSL was equal to 3.1 MECU.

Public health damages are a consequence of the impacts of air emissions: the emissions from the stack of the power plant and the air emissions from all the other stages which increase mortality (VOLY) and morbidity. In addition, health impacts of road accidents caused by the truck transport of peat in Finland were included in these figures. Only the VSL principle was applied to the accidents.

The damages of occupational accidents and diseases in Finnish power plants were assumed to be a negligible factor.

The power generation stage is responsible for the major part of the health damages of the peat fuel cycle: its share seems to be over 90% using the VSL principle. Only VSL approach was applied to the accidents, so that damage comparisons within the whole peat cycle cannot be made using VOLY.

5.4.2. Global warming

Global warming is probably the most important damage factor of the Finnish peat cycle when applying the ExternE methodology. The power generation stage causes over 90% of the greenhouse impacts of the whole cycle (when the using the GWP_{100} coefficients for the greenhouse gases). The restoration stage of the peat cycle includes an interesting view of the sink effect of the CO_2 emissions. During the time span of 100 years the sink effect of the restoration stage is about 10%. The sink effect increases if the time period increases. In the rewetting mode the time period of about 2000 years is long enough to abolish the CO_2 emissions generated in the power generation stage.

5.4.3. Other damages evaluated

The impacts on crops via ozone seem to be the most important of the quantified damages.

Hitherto, no monetary measures could be developed in the ExternE Project for ecosystem damages. Only the impacts on ecosystems were quantified in the form of increase in land area where the critical load of acidity was exceeded. The results are summarised in Table 1.3 in

Appendix IX. It should be remembered that all land areas to the east of Finland are here excluded.

The valuation of the material damages is based on the database of EcoSense, which includes the surface areas of different kinds of building materials and their monetary values. The building materials are exposed by the acid deposit, which causes the damages. Road damages caused by transport were not assessed in the Finnish implementation.

5.5. Interpretation of the results and sensitivity analysis

The damage estimates in the ExternE Project are dominated by health impacts (especially mortal ones) and global warming. Consequently most of the theoretical work in the ExternE Project is concentrated on human health impacts. The impacts on ecosystems seem to be very difficult to assess, particularly the long-term impacts. It is even more difficult to give any monetary valuation for this kind of damages. The evaluation of material damages could be realistic if the database of EcoSense included reliable data on building materials in Europe, but this remained an open question to the Finnish team (as well as the data concerning natural ecosystems). These damages are still small compared to the human health impacts.

The truck transports account for only a negligible part of the total accidents. The occupational accidents in the Finnish peat power plant and peat production were not valued.

The EcoSense model allows a calculation of the TSP, SO_2 and NO_x impacts and damages in Western Europe by taking into account the dispersion and chemical reactions in the air. The calculation of ozone damage is at present much coarser and the results an order of magnitude more uncertain. The same damage estimate ECU/t (proportional to the emitted amount of NO_x) was applied all over Europe, which may cause an overestimate of damages caused by Finnish power plants. Therefore the ozone damages to crops seem totally to dominate over the SO_2 damages in the calculations.

The estimates of global warming are an order of magnitude more uncertain. Although the emissions of greenhouse gases and their global warming potentials (GWPs) are well known, knowledge of the true impacts and damages of GW is still poor.

The approximation that the pollutants from other stages also emit through the stack probably slightly underestimates the impacts of the emissions, which in reality occur in more densely populated areas and closer to the ground.

Table 5.6 Damages of the peat fuel cycle (north-west Russian population excluded).

	mECU/kWh	σ_g
POWER GENERATION		
Public health		
Mortality*- VOLY *(VSL)*	3.1 *(10)*	B
of which TSP	*0.15 (0.54)*	
SO_2	*1.65 (6.0)*	
NO_x	*0.83 (3.0)*	
NO_x (via ozone)	*0.45*	
Morbidity	1.1	
of which TSP, SO_2, NO_x	*0.3*	A
NO_x (via ozone)	*0.8*	B
Accidents	nq	A
Occupational health	nq	A
Crops	0.4	B
of which SO_2	*0.007*	
NO_x (via ozone)	*0.4*	
Ecosystems	iq	B
Materials	0.09	B
Monuments	nq	
Noise	nq	
Visual impacts	nq	
Global warming		C
low	2.9	
mid 3%	16	
mid 1%	45	
high	135	
OTHER FUEL CYCLE STAGES		
Public health		
Outside EU		B
Inside EU	*0.67(0.96)*	B
Occupational health		A
Outside EU		
Inside EU	*nq*	
Ecological effects	nq	B
Road damages	nq	A
Global warming		C
low	0.11	
mid 3%	0.56	
mid 1%	1.6	
high	4.9	

*VOLY= mortality impacts based on 'value of life year' approach, VSL= impacts evaluated based on 'value of statistical life' approach; σ_g = standard deviation confidence band (see Appendix VII);
ng: negligible; nq: not quantified; iq: only impact quantified; - : not relevant

Table 5.7 Sub-total damages of the peat fuel cycle.

		mECU/kWh
VOLY *(VSL)*	low	nq
mid		**5.4 *(12.6)***
upper		nq

Table 5.8 Damages by pollutant.

	ECU / t of pollutant
SO_2 *- VOLY *(VSL)*	1027 *(3310)*
NO_x *- VOLY *(VSL)*	856 *(2886)*
TSP *- VOLY *(VSL)*	1340 *(4528)*
NO_x (via ozone)	1500
CO_2	3—139

*VOLY= mortality impacts based on 'years of life lost' approach, (VSL= impacts evaluated based on 'value of statistical life' approach.)

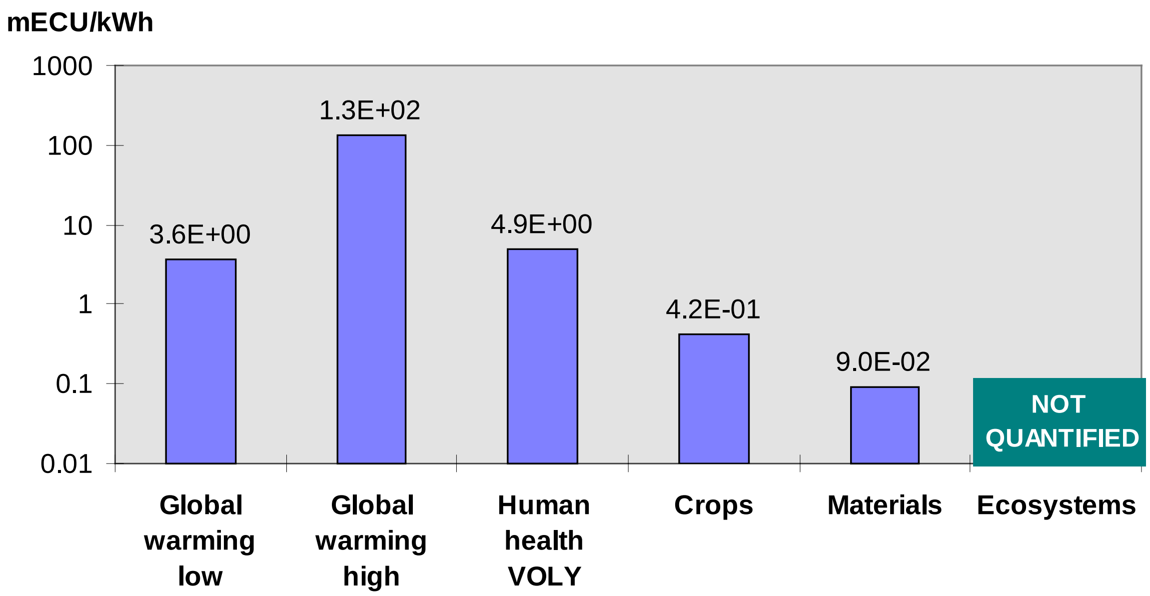


Figure 5.1 Total damages of the peat fuel cycle by impact category.

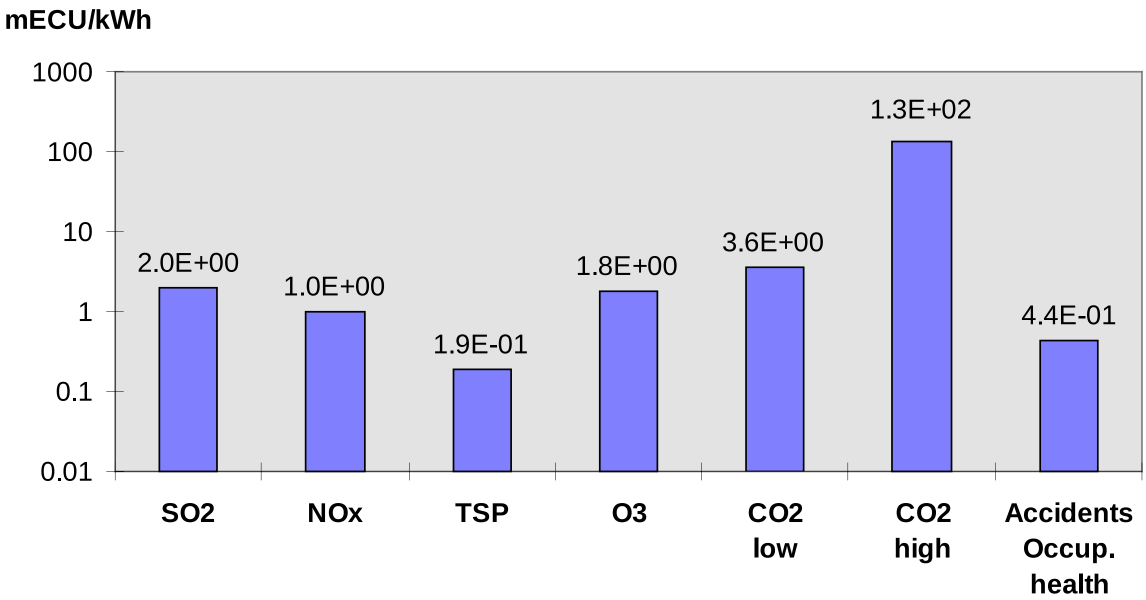


Figure 5.2 Total damages of the peat fuel cycle by burden category.

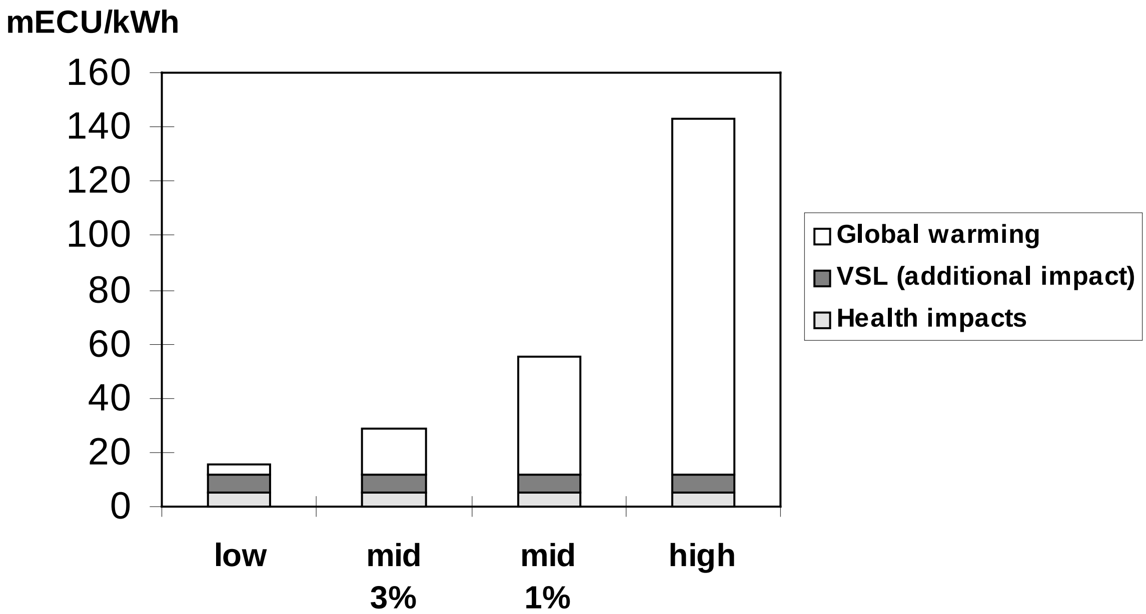


Figure 5.3 Comparison of different GW damage estimates and health damage estimates for the peat fuel cycle.

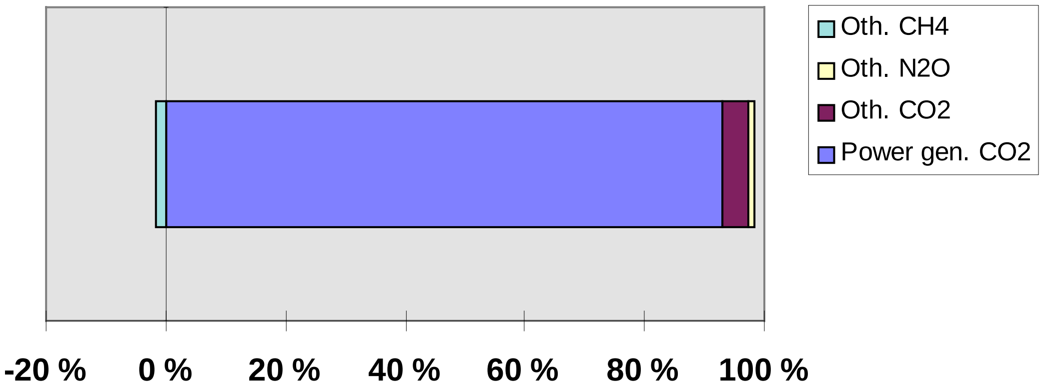


Figure 5.4 Share of the different GHG emissions in the global warming impact of the peat fuel cycle.

One problem concerning the Finnish national implementation were the limits of the computational grid in EcoSense (Finland being situated at one edge of the grid), which was assumed to underestimate the real damages. As the prevailing winds come from the south west, the emission impacts outside the Finnish borders take place mainly on the north eastern side of the country. The affected Russian population is not considered in the peat fuel cycle.

In the Finnish coal fuel cycle the sensitivity of the damage estimates to this additional population of 8.4 million Russian people was investigated. After adding the Russian population the damages of SO_2 and TSP of the coal fuel cycle were increased by 17% and the damages of NO_x by 31%. This indicates clearly how the impacts of NO_x emissions extend further than those of the other two pollutant components.

6. Biomass fuel cycle

6.1. Definition of the fuel cycle

The plant chosen to represent the biomass fuel cycle is a new plant located in the town of Forssa. It began operation in autumn 1996. The thermal power of the combined heat and power (CHP) generation plant is 72 MW. It is the first district heat and electricity producing plant of this size using solely wood biomass as fuel. Finland has high potential in the utilisation of renewable domestic energy resources, especially bioenergy. The forests are the main potential source for bioenergy in Finland. Consequently this plant might represent a typical example of future environmentally more acceptable energy technology in Finland. The stages of the biomass fuel cycle are shown in Figure 6.1. A detailed definition of the Finnish biomass fuel cycle is given in Appendix X, where the burdens, impacts and damage figures of the fuel cycle are also listed.

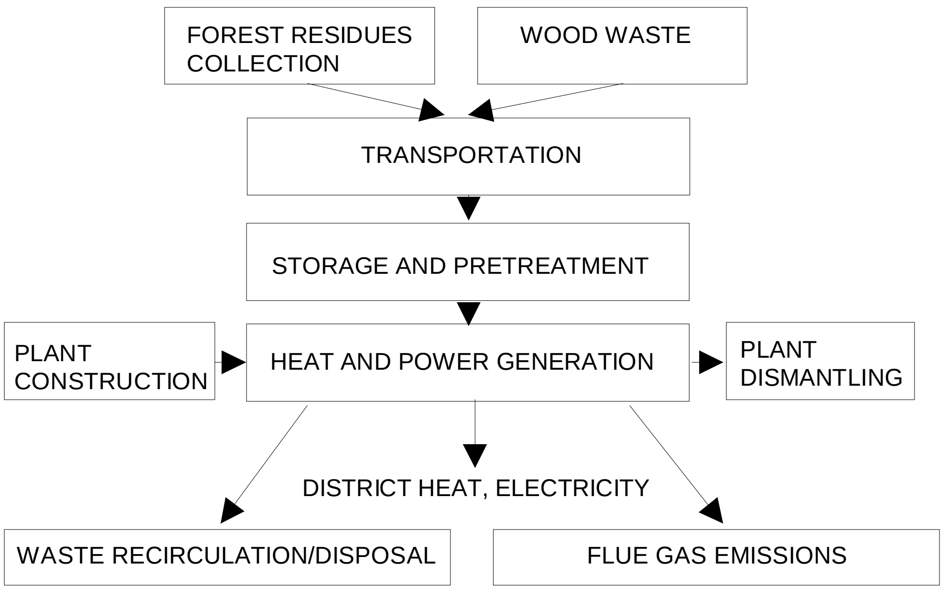


Figure 6.1 Stages of the biomass fuel cycle.

Biomass is typically a local fuel, available in a limited supply within cost-effective transport distance. The scaling down of proven technology and plant concepts makes possible a significant increase in the utilisation of biomass in energy production. Typically a number of factors and features combine to make small-scale energy production from biomass feasible.

Simultaneous production of electricity and district heat or prosess steam boots the profitability of small scale power plants. High total plant efficiency is an important feature, as well as the capability to burn a wide variety of fuels.

6.1.1. Power generation stage

The power specifications of the Forssa CHP plant are: 48MW_h heat and 17.2 MW_e electricity. The supplier of the bubbling fluidised bed (BFB) boiler was Foster Wheeler. The boiler output is 66 MW_{th}, the steam temperature 510 ^oC and the pressure 61 bar. Because the plant was new, started in August-September 1996, when performing this case study the energy production and the emissions were estimated from the theoretical specifications of the power station.

The fuel mix consists of saw dust, bark and wood waste. The average values of wood fuel parameters are shown in the Table 6.1.

Table 6.1 Average values of wood fuel parameters.

Moisture content	40—55%
Ash content (% of dry matter)	0.4—2.8%
Calorific value (MJ/kg, dry matter)	19—20 MJ/kg
Sulphur content (% of dry matter)	about 0.05%

The electricity/heat rate of the plant is 0.263 at full effect and 0.288 at 40% effect. The district heat pipe line to the town is 3.5 km long. The plant produce 95% of the district heat needed in the town. One third of the electric power the company Forssan Energia send out to the power distribution network.

Flue gases are cleaned with an electrostatic precipitator. The technical data for the cleaned flue gases are shown in the following table. (1 mg/MJ $\cong$ 2.5 mg/ Nm^3). The flue gas volume is 30 Nm^3/s, temperature 130^oC (403 K) the stack height 50 m, and stack diameter 1.6 m. SO_2, NO_x, dust and CO are measured continuously at the plant.

6.1.2. Upstream fuel-cycle operations

The fuel mix consists of saw dust, bark and wood waste. Almost 80% of the fuel is produced as a by product from saw mills. Slightly more than 10% of the fuel comes directly from the forest and less than 10% consists of other kinds of waste wood. The fuel chips coming directly from forest land are transported 0—50 km and other wood waste fuels up to about 100 km using empty transportation vehicles (return transport). The different wood fuels are brought from more than 20 different places situated 0—100 km from the plant. Of the approximately 400 000 i-m^3 wood fuel needed, or about 4 000 trucks per year, the estimated for the calculated reference year was: 130 000 m^3 saw dust, 185 000 m^3 bark, 50 000 m^3 wood chips direct from the forest and less than 35 000 m^3 other waste wood fractions.

The principle of the saw mill process is described in. Figure 6.2. Saw mills produce fuel for the Forssa plant. The use of round wood and the calculated amount of by products are shown in Table 6.2. It should be noted that the saw mills use some of the produced waste wood themselves for the drying process. The wood waste produced at the saw mills is also used in other energy producing facilities.

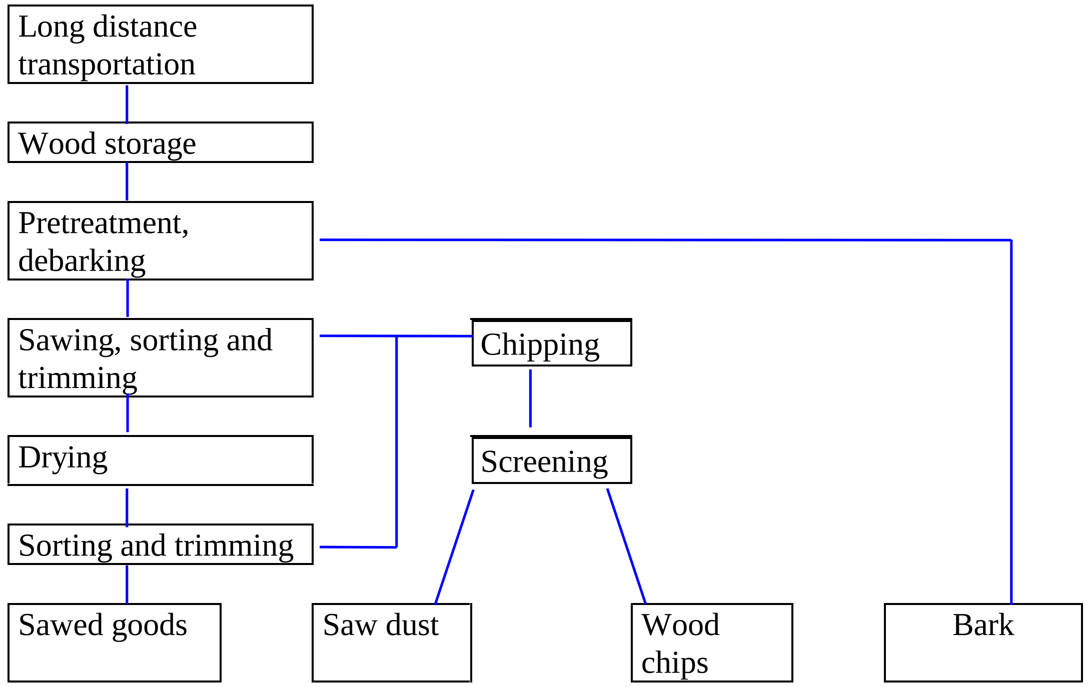


Figure 6.2 Process scheme of a typical saw mill, main and by-products.

The process scheme of raw material wood and fuel chips from forestry land is presented in Figure 6.3. Energy needed for production of wood chips with the chipset-system is 1.08 l diesel fuel/m^3 chips (1.35 l/MWh wood fuel) and the electricity needed in the sawmill chain 2.56 kWh/MWh wood fuel. For the transportation a diesel fuel consumption of 51 l /100 km is used.

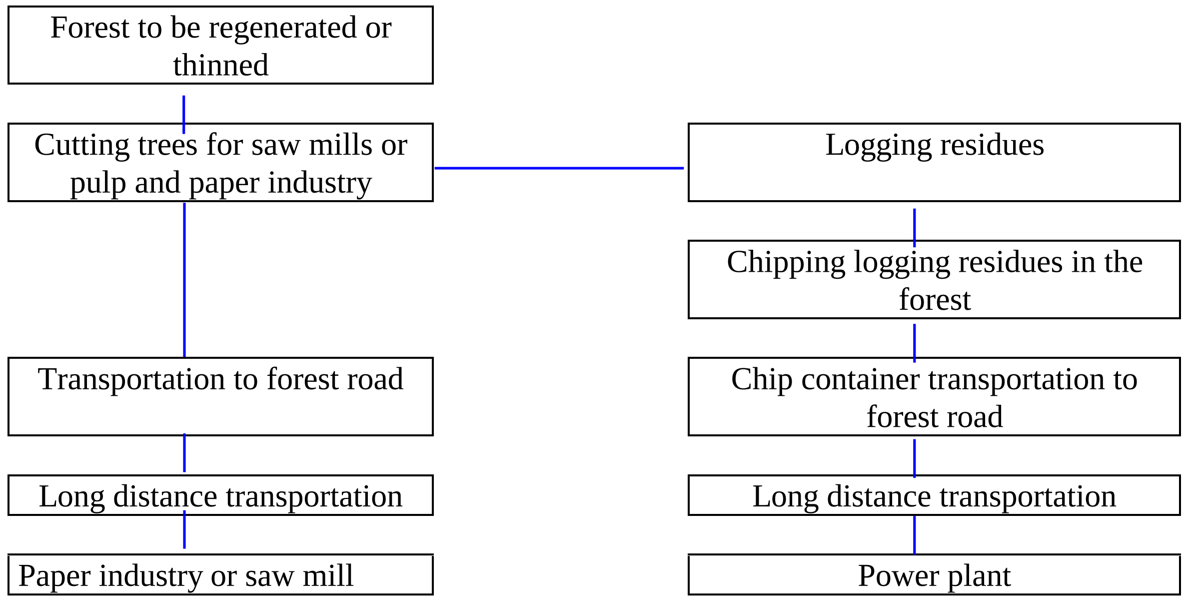


Figure 6.3 Process scheme of raw material wood and fuel chips from forestry land.

Table 6.2 Available saw mill fuels for the Forssa plant.

Saw mill	Distance from Forssa, km	Round wood m^3	Birch(B) Pine(P) Spruce(S)	Main product m^3	Bark loose cubic metre	Saw dust loose cubic metre	Wood chips loose cubic metre
Metsä-Serla Oy, Kyrö	ca 40 km	320 000	P 65% B 35%	140 000	96 000	101 000	199 000
Oy Metsä Timber Ltd, Rengo	ca 40 km	470 000	S 100%	200 000	141 000	148 000	292 000
Asko Oy, Forssa saw mill	3 km	100 000	S 70% P 30%	45 000	30 000	31 000	62 000
Raunila saw mill, Koski plant	40 km	220 000	S 100%	100 000	66 000	69 000	137 000
Humppila saw mill	20 km	80 000	S 75% P 25%	35 000	24 000	25 000	50 000

The main fuel storage at the plant has four departments and can take 800 m^3 of fuel. There are also two boiler silos, each holding 150 m^3. Most of the fuel is stored in stockpiles in the open. This is a normal situation in Finland, where the fuel needed is usually brought to the plant continuously. The result of storage has been that the moisture content of the base fuel during the first season was higher than planned.

6.1.3. Downstream fuel-cycle operations

The generation of wood ashes is 500—1000 t/a. Ashes and other wastes are stored at the waste disposal area near to the plant. Wood ashes can be used as construction material in the waste disposal area.

The construction and demolition of facilities was not taken into account in this study. The planned lifetime for this kind of facility is normally 25—30 years, but in practice parts of the plant can probably be modernised so that the real lifetime will be much longer. The construction period lasted about 1.5 years and the building costs (budget) were 105 million FIM (about 18 MECU).

6.1.4. Site description

The plant is situated in southern Finland, in the province of Häme (see Figure 1.1) near the border to the province of Turku and Pori and the province of Uusimaa. The population density of these provinces (35—131 inhabitants/km^2) is greater than on average in Finland (17 inhabitants/km^2).

The exact location of the power station is about 3 km west of the town centre of Forssa, with 20 000 inhabitants. The geographical coordinates of the plant are: latitude 60.79°, longitude 23.59° (east). Most of the area is situated less than 125 m above sea level. The land area is also used for waste management activities (dump under construction). The nearest settlement is situated 1 km from the plant. The nearest recreation area is also about 1 km from the plant and the nearest protected area (Torronsuo) 2.5 km away. The power plant area and the surrounding land have no special environmental protection value.

The air quality in Forssa is most dependent on traffic emissions and sulphur emissions from the use of oil. As the sulphur content in the oil used has been restricted not to exceed 1%, the sulphur emissions have radically decreased since the 1980s. The emissions from traffic and other energy production facilities in the town of Forssa have been estimated to be about 500 t NO_x, 500 t SO_2 and 270 t dust. The development of air quality has been studied by the use of biological indicators (lichen) since the 1980s.

The upstream parts of the fuel cycle also lie in the neighbourhood of Forssa, as described in chapter 6.1.2.

6.2. Overview of burdens

6.2.1. Air emissions

The major burdens of the biomass fuel cycle are the atmospheric emissions of pollutants from the power generation stage. The major air pollutants are SO_2, NO_x, TSP and N_2O. The CO_2 emissions from burning biomass are not taken into account as global warming impacts because the forestry is on a sustainable basis. However, the burning causes emissions of N_2O, which is a powerful greenhouse gas. The small fossil CO_2 emissions come from transport and production of wood fuel and from light fuel oil used as booster fuel in the boiler.

The specific emissions of the power generation stage are given in Table 6.3. These numbers present upper limits of the emissions and were used in the EcoSense runs. The EcoSense input data for the plant is given in the Table 6.4.

Table 6.3. Air emissions of the power generation stage used in the model runs and their percentage of the whole fuel cycle emissions.

	Specific emissions g/kWh_e	Specific emissions mg/MJ_{fuel}	Emissions t /a	% of the whole biomass fuel cycle
SO_2	0.41	40*)	37	97
NO_x	1.56	150*)	140	93
TSP	0.21	20	19	96
fossil CO_2	1.25	121	113**)	15
CH_4	0.41	40	37	99.7
N_2O	0.21	20	19***)	99.8

*)according to recent measurements the SO_2 emissions are less than 5 mg/MJ and NO_x emissions 80—90 mg/MJ.
**)from light fuel oil, 113 000 t CO_2 from biomass, which is a part of the natural carbon cycle and considered not to impact on global warming.
***)the actual emissions may be only 1/10.

Table 6.4 EcoSense input data for the Forssa biomass plant.

Full load hours per year	3600 h
Stack hight	50 m
Stack diameter	1.6 m
SO_2 Emission	96.0 mg/Nm^3
NO_x Emission	360 mg/Nm^3
TSP Emission	48 mg/Nm^3
Flue gas volume stream	108 000 Nm^3/h
Flue gas temperature	403 K
Surface elevation at site	128 m

Total emissions from fuel production and transportation are shown in Table 6.5.

Table 6.5 Total emissions from production and transportation for the different fuel alternatives.

Total emissions		CO t	NO_x t	Part. t	SO2 t	CO2 1000 t
Chipset	prod.	0.81	2.70	0.27	0.22	0.15
	transp.	0.17	0.96	0.06	0.07	0.05
Saw mil bark	prod.	0.27	0.22	0.05	0.22	0.09
	transp.	0.62	3.57	0.21	0.27	0.19
Saw mill dust	prod.	0.00	0.00	0.00	0.00	0.00
	transp.	0.43	2.51	0.14	0.19	0.13
Other wood fuels	transp.	0.15	0.84	0.05	0.06	0.04
Production total		1.08	2.92	0.32	0.43	0.24
Transportation total		1.36	7.88	0.45	0.61	0.41

As can be seen the main part of the emissions to the air is caused by the power generation stage with the exception of carbon dioxide emissions.

Outside the combustion stage, the SO_2, CO, TSP and CO_2 emissions originate from both fuel production and transportation (almost fifty-fifty), whereas the different transportation stages are responsible for most of the NO_x emissions.

Emissions (and their impacts) from power plant construction and demolition were not estimated in the Finnish national implementation. According to the previous ExternE results (EC, 1995) these impacts are small, at least for fossil fuel cycles, compared to the power generation stage during the life-time of the plant. An assessment of this issue might be needed in the wood biomass fuel cycle.

6.2.2. Waste water and solid waste

Process water, cooling water and water for social purposes are taken from the communal water supply, about 150 m^3/day, and after use sent out to the municipal sewage network. As the wastewater must fulfil the municipal standards the environmental effects are assumed to be negligible.

The amount of biomass ash from the plant is small, about 5000 t /a. It will be used for landfill purposes nearby the plant. Later on recirculation to the forest may be initiated. The environmental effects of ash disposal are assumed to be negligible.

6.2.3. Direct public and occupational health effects

Accidents during biomass production and transportation can approximately be estimated upon from accident frequency figures for forest work (the whole sector) and normal road traffic. The frequency for forest work has been between 2E-5 and 3E-5 injuries per MWh produced from biomass during the period 1990-1995. The frequency for fatal accidents was 5E-8 injuries per MWh produced biomass. Road traffic accident numbers for Finland during the above period were 19—32 persons injured and 1.0—1.6 persons killed per 100 million km. For Forssa with an annual transportation need of 300 000 km and 260 GWh this means a frequency of 2E-7—3E-7 injuries and about E-8 deaths per MWh fuel transportated.

6.2.4. Noise

The noise level does not exceed 45 dB (A) at the border of the power plant area. The value does not include safety valves. Noise disturbances from incoming and outgoing traffic and from service machines may be identified but they are here assumed to be negligible.

6.3. Selection of priority impacts

A list of the principal impacts of the biomass fuel cycle is given in Table 6.6.

Table 6.6 Principal impacts of the biomass fuel cycle.

Impacts	Production	Transport	Power generation	Waste disposal/ recycling	Construction/ demolition
Global warming	x	x	x		x
Public health		x	x	x	x
Occupational health	x	x	x		x
Crops	x	x	x		x
Forests	x	x	x	x	x
Ecosystems	x	x	x	x	x
Materials		x	x		x
Noise	x	x	x		x
Road traffic		x		x	x
Visual impact	x		x	x	x

According to previous studies of the ExternE Project (EC, 1995) and the experiences of the National Implementation projects the most important impacts of the biomass fuel cycle are those caused by atmospheric emissions, especially from the power generation stage.

The impacts identified as primary for the Finnish biomass fuel cycle were:

- global warming due to release of greenhouse gases during fuel production, transportation and combustion
- effects of atmospheric pollution on human health
- injuries and occupational health risks to the workers and the general public
- effects of air pollution and acidic deposition on the built-up environment
- effects of air pollution on crops
- effects of air pollution and acidic deposition on natural ecosystems
- impacts of biomass removal on the ecosystem

Because of lack of quantitative information on impacts of biomass removal on the ecosystems the last impact group could not be assessed in this study.

6.4. Quantification of impacts and damages

For general comments, see chapter 4.4. Note that the Russian health impacts have not been taken into account in the calculations for the biomass fuel cycle.

Because the power station produces both electricity and heat, the burdens caused by electricity production are not unambiguous. The allocation has here been performed by the exergy principle, which causes about twofold impacts and damages compared with allocation performed by the energy principle.

The damages caused by emissions from fuel production and transportation have not been calculated by any model but in order to get an idea of their magnitude in proportion to damages caused by emissions from the power plant the production and transportation emissions were here studied as if they had come from the stack of the power plant.

In the following the impacts and their damage estimates are discussed by impact category. The summarised estimates for the biomass fuel chain are given in Table 6.7**,** Table 6.8 and Table 6.9 and Figure 6.4, Figure 6.5, Figure 6.6 and Figure 6.7. More detailed results are presented in Table X.3 of Appendix X.

As noted earlier, some impact categories, especially impacts on ecosystems, have not been valuated in monetary terms.

6.4.1. Public and occupational health

For general comments, see chapter 4.4.1. The main damage is caused to human health. The power generation stage is responsible for the major part of the health damages of the biomass fuel cycle: its share seems to be 85%. Another important damage factor of the Finnish biomass fuel cycle are the presumable biomass production and transportation accidents. An open methodological question is further whether the costs of these accidents are covered by employer and vehicle insurance systems internalised in the costs of the fuel.

6.4.2. Global warming

Global warming is only a small damage factor in the biomass fuel cycle compared with the coal and peat fuel cycles. The estimated GW damages for the biomass cycle seem to be less than 10% of those of the coal chain. The power generation stage causes 80% of the greenhouse impacts of the whole biomass fuel cycle (when using the GWP_{100} coefficients for the greenhouse gases).

6.4.3. Other damages evaluated

For general comments, see section 4.4.3. No monetary measures could yet be developed in the ExternE Project for ecosystem damages. Only the *impacts* on the ecosystem where quantified in the form of increase in land area where critical load of acidity was exceeded. The summarised results are given in Table X.3 in Appendix X. It is questionable whether the sulphur content in biomass should be considered a burden causing impacts to the ecosystem as it is in any case a part of the natural sulphur cycle The removal of biomass removes approximately the same amount of sulphur as is returned with the flue gases from biomass. Note that the areas of distribution are not the same.

Table 6.7 Damages of the biomass fuel cycle (north-west Russian population excluded).

	mECU/kWh	σ_g
POWER GENERATION		
Public health		
Mortality*- VOLY *(VSL)*	3.6 *(11.4)*	B
of which TSP	0.48 *(1.8)*	
SO_2	0.55 *(2.0)*	
NO_x	1.9 *(7.0)*	
NO_x (via ozone)	0.65	
Morbidity	1.5	
of which TSP, SO_2, NO_x	0.38	A
NO_x (via ozone)	1.1	B
Accidents	nq	
Occupational health	nq	
Major accidents	-	
Crops	0.55	B
of which SO_2	0.003	
NO_x (via ozone)	0.54	
Ecosystems	iq	B
Materials	0.05	B
Monuments	nq	
Noise	nq	
Visual impacts	nq	
Global warming		C
low	0.22	
mid 3%	1.2	
mid 1%	3.4	
high	10.3	
OTHER FUEL CYCLE STAGES		
Public health		
Outside EU	-	*B*
Inside EU	0.80 *(1.3)*	*B*
Occupational health		
Outside EU	-	*A*
Inside EU	0.11 *(0.11)*	
Ecological effects	nq	
Global warming		C
low	0.02	
mid 3%	0.12	
mid	0.34	
high	1.0	

*VOLY= mortality impacts based on ‘value of life year’ approach, VSL= impacts evaluated based on ‘value of statistical life’ approach; σ_g = standard deviation confidence band (see Appendix VII);
ng: negligible; nq: not quantified; iq: only impact quantified; - : not relevant

Table 6.8 Total health damages of the biomass fuel cycle **(***north-west Russian population excluded)***.**

		mECU/kWh
VOLY *(VSL)*	low	nq
	mid	6.0 *(14.3)*
	upper	nq

Table 6.9 Total quantified damages of the biomass fuel cycle per tonne of pollutant **(***north-west Russian population excluded, no ecosystems)***.**

	ECU / t of pollutant
SO_2 *- VOLY *(VSL)*	1606 *(5055)*
NO_x *- VOLY *(VSL)*	1388 *(4679)*
TSP *- VOLY *(VSL)*	2611 *(8800)*
NO_x (via ozone)	1500
CO_2	3—139

*VOLY= mortality impacts based on 'years of life lost' approach, (VSL= impacts evaluated based on 'value of statistical life' approach.)

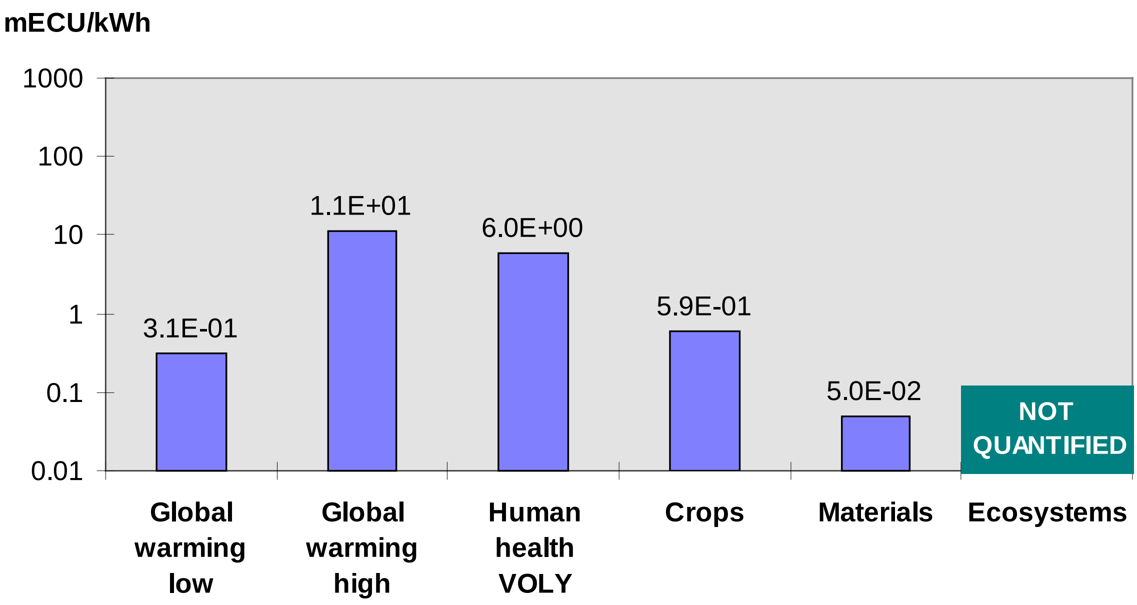


Figure 6.4 Total damages of the biomass fuel cycle by impact category.

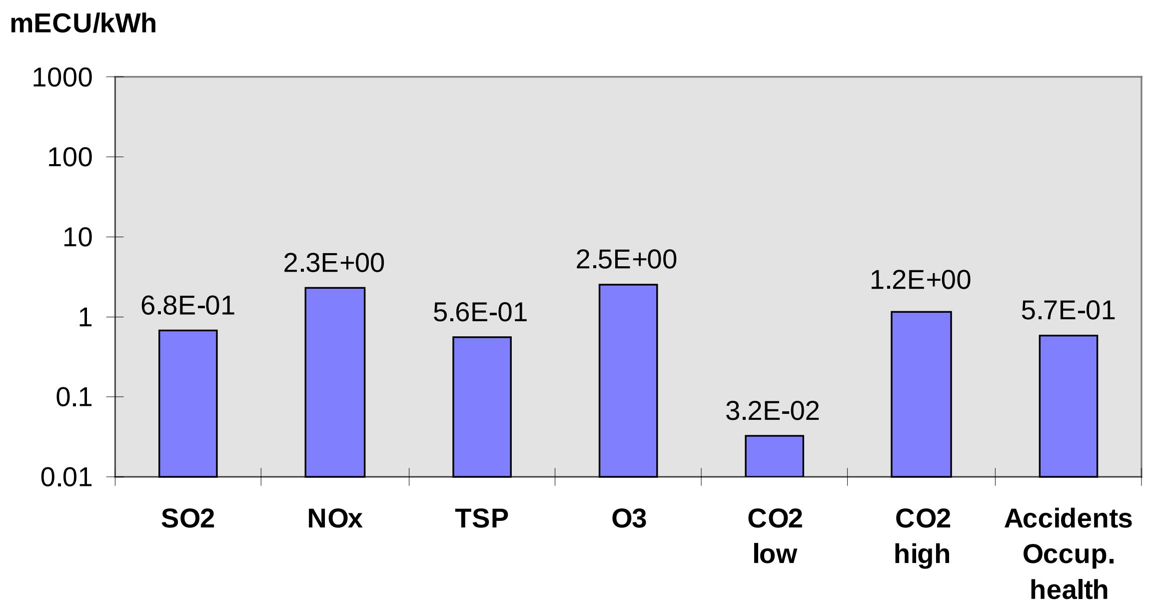


Figure 6.5 Total damages of the biomass fuel cycle by burden category.

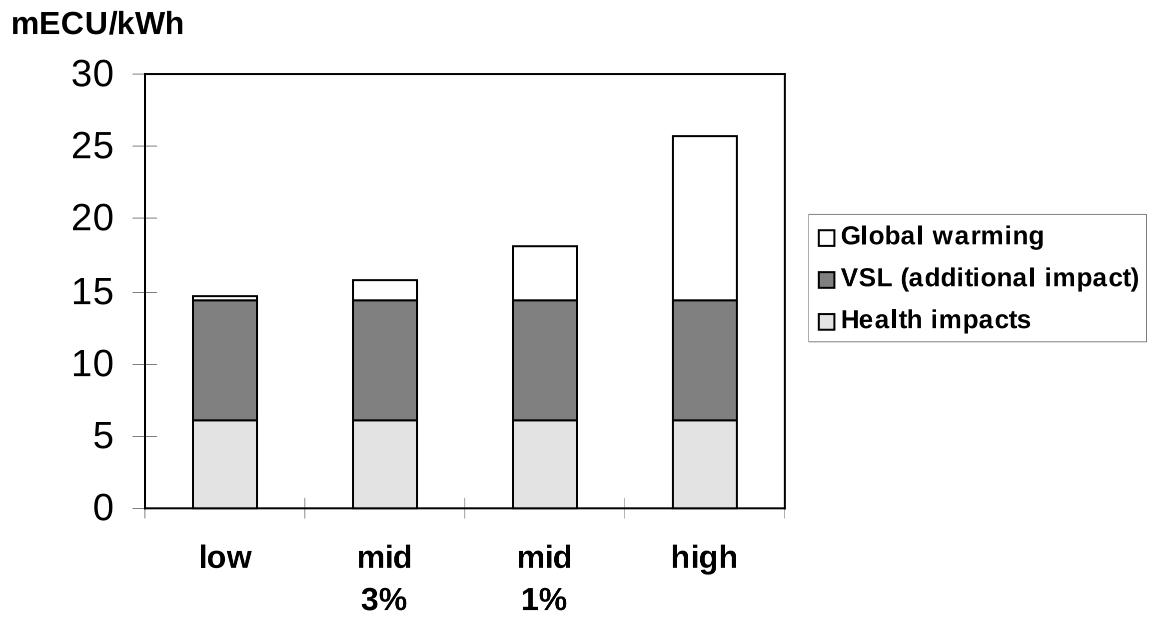


Figure 6.6 Comparison of different GW damage estimates and health damage estimates for the biomass fuel cycle.

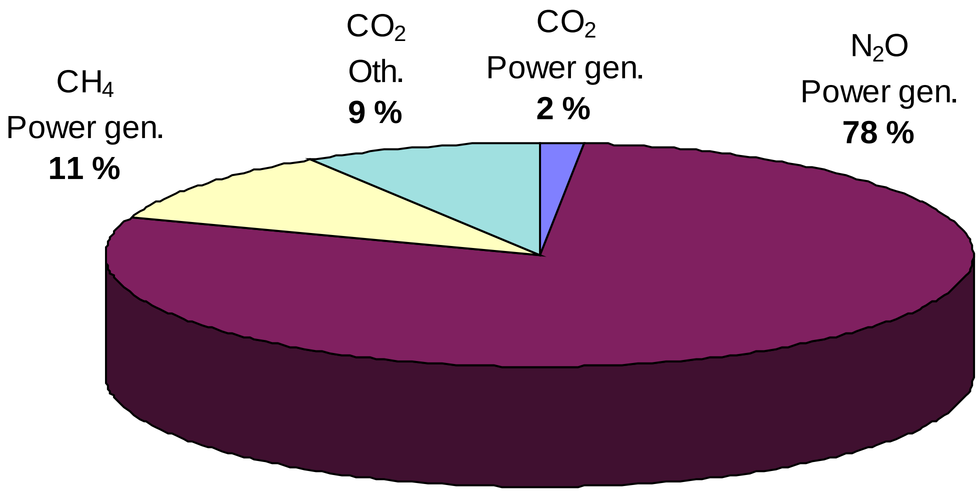


Figure 6.7 Share of the different GHG emissions in the global warming impact of the biomass fuel cycle.

6.5. Interpretation of the results and sensitivity analysis

For general comments on the calculation methods, see chapter 4.5. The greater damage in the case of biomass is caused to human health.

According to later measurements from the plant the emissions of the power generation stage, particularly SO_2 and N_2O, were overestimated in the calculations. Hence as well the GW and health impacts are overestimated in the results presented.

The approximation that the pollutants (due to combustion) from fuel production and transportation are emitted through the power plant stack probably slightly underestimates the impacts of the emissions, which in reality occur closer to the ground.

The calculated results for the biomass chain do not include figures for areas east of Finland. It is possible to get an idea of the magnitude of the missing figures by examining the Finnish coal chain, in which Russian health impacts have been considered. The sensitivity of the damage estimates to this additional population of 8.4 million people was calculated to be relatively small in the coal chain. The damages of SO_2 and TSP were increased by 17% and the damages of NO_x by 31%. As the stack at the coal plant is three times as high as at the biomass plant it can be assumed that the relative impacts on the Russian area are smaller for the biomass cycle than for the coal cycle.

7. Aggregation

7.1. Comparison of results between fuel cycles

The three fuel cycles and their estimated impacts were described in detail in Chapters 4, 5 and 6. Here comparative notes on some particular results are given.

The specific emissions of the fuel cycles considered in the ExternE National Implementation Project of Finland are summarised in Table 7.1. The estimated damages caused by the airborne emissions from the corresponding fuel cycles are summarised in Table 7.2 (section 7.4).

The damage results are not fully comparable with each other because the health impacts in Russia (in areas close to Finnish borders) were taken into account only in the coal fuel cycle and the locations of the plants differ. If these were also considered in the peat and biomass cycles, their damage numbers attributed to sulphur, nitrogen and particulate emissions would probably be about 15—30% larger.

In the case of the coal fuel cycle the damages caused by SO_2 and TSP were increased by 17% and those due to NO_x by 31% if the above mentioned Russian population of 8.4 million were added to the data base of the model. Because the stacks of the peat and biomass power plants are shorter than in the case of coal fuel cycle, their additional health damages in Russia are probably smaller.

The different structures of the power plants create another problem which makes comparison of the fuel cycles more difficult. In the peat and biomass fuel cycles the plants are of CHP type, which means that the estimated impacts and damages must be allocated between electricity and heat generation. In the case of the coal fuel cycle, in which electricity is the only product of the condensing plant there are no allocation problems.

In the ExternE National Implementation it was decided to use the exergy of the power generated as an allocation principle of impacts and damages. First the total damages of the whole fuel cycle were estimated and then the total damages were allocated between electricity and heat. This means in practice that a great majority of the total damage is assigned to electricity. Consequently, in this study in which only electricity generation is considered the energy efficiency of CHP is not highlighted. An illustrative example is the peat fired plant: a condensing plant with an efficiency slightly more than 41% would give better damage figures for electricity generation than the CHP plant considered in this NI study with a total energy efficiency of 86%.

Another possible way of considering the heat production would be to augment the system boundaries of the fuel cycle (see Schlamadinger et al. 1997) and to include consideration of the district heating system. In the reference case (to which the fuel chain is compared) the same amount of heat would be generated separately. The net change of impacts/damages compared to the reference case would then give the figures for the CHP case.

Table 7.1 Specific emissions of SO_2, NO_x and TSP of the National Implementation case plants.

	Power generation g/kWh$_e$	Other stages g/kWh$_e$	Whole cycle g/kWh$_e$
COAL Meri-Pori			
SO_2	0.67	0.16	0.83
NO_x	0.53	0.10	0.63
TSP	0.15	0.02	0.17
CO_2	770	25	795
PEAT Rauhalahti (CHP)*			
SO_2	1.90	0.03	1.93
NO_x	1.09	0.09	1.18
TSP	0.12	0.01	0.14
CO_2	900	40	940
BIOMASS Forssa (CHP)*			
SO_2	0.41	0.01	0.43
NO_x	1.56	0.12	1.68
TSP	0.21	0.01	0.22
CO_2	1.3	7.2	8.5

* burdens allocated to electricity, exergy principle applied

The estimated GW damages caused by greenhouse gases are the most important for the two fossil fuel cycles, coal and peat (Table 7.2). Peat is worse than coal due to its higher specific emissions of CO_2 (i.e. emissions per primary energy content). A small reduction in greenhouse impacts of peat is caused by the decrease of CH_4 emissions from peatlands after the change in land use change. In the biomass fuel cycle the small fossil CO_2 emissions (causing global warming) originate from the production of fuel and its transportation to the power plant. The greatest greenhouse impacts in the biomass fuel cycle are caused by the N_2O emissions from the fluidised bed boiler, but these are still negligible compared to the fossil fuels.

The damages caused to human health dominate totally the valuated impacts of sulphur, nitrogen and particulate emissions in all the fuel cycles. However, these damages appear to be essentially smaller in Finland than in the Central European National Implementation studies, mainly due to the sparse population in Finland. Consequently these results could likely be misused in an imaginable damage minimisation strategy inside EU: An inconsiderate conclusion would be to relocate the power plants in Northern Europe and then sell the electricity to the crowded areas in Central Europe!

7.2. Quantified description of the national electricity sector

The Finnish electricity supply system includes almost 400 power stations with a total generation capacity of about 14 000 MW (1996). Small hydro or CHP plants are greatest in number, but the 10 largest plants (including four nuclear plant units) account for about 40% of the total capacity. Excluding the nuclear plants, the largest power plant is the Meri-Pori station commissioned at the end of 1993. The plant has a net capacity of 560 MW_e, and a net electric efficiency of 41–42%.

The total net supply of electricity was about 69 TWh in 1995. The structure of the electricity supply is presented in Figure 7.1 by energy source both for Finland and for the European Union as a whole. In general, the differences between Finland and the average of EU countries are relatively small. However, the scale of biomass use for electricity generation is particularly large in Finland compared to other EU countries. In 1994 biomass accounted for about 10% of the total output of electricity generation, whereas the average within EU was only about 1%. The share of peat-based electricity generation was about 7.5% in 1994. Oil has at present only a very small role in the Finnish electricity generation system.

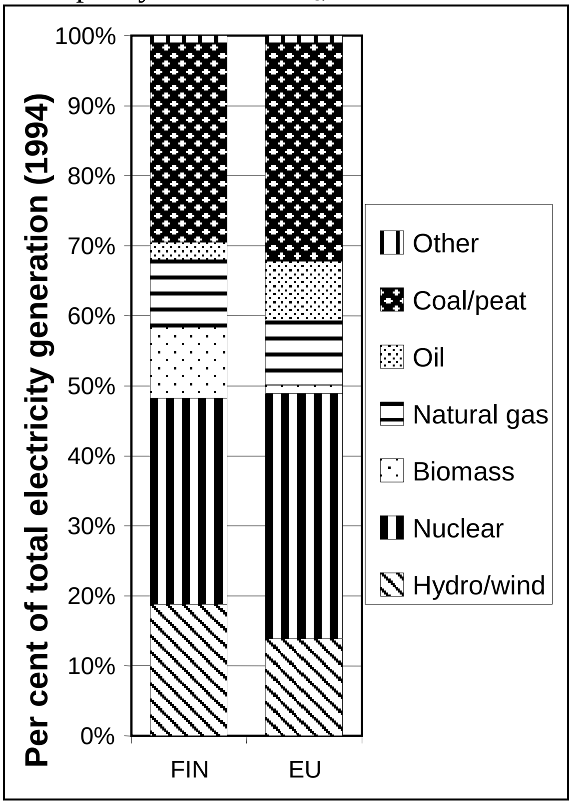


Figure 7.1 Structure of total electricity generation according to energy source in Finland and the European Union.

The structure of electricity supply and consumption is illustrated in Figure 7.2 by type of generation and by main consumption category. An important feature of the Finnish system is the large share of CHP in the overall electricity supply. District heat CHP accounted for over 16%, and industrial CHP for about 14% of the total generation in 1995. Consequently, the average efficiency of fuel-based electricity generation is considerably higher in Finland than on average within the European Union. For the year 1994, the average efficiency has been estimated as about 57% (Lehtilä et al. 1997).

On per capita terms, the consumption of electricity in Finland was about 13.5 MWh in 1994, which was more than double compared to the average within EU (5.8 MWh). As to the structure of consumption, particularly the industrial use of electricity is very high in Finland compared to the average within EU. In this respect Finland is much more similar to Sweden and Norway than to the EU average, although in those two countries the industrial consumption is on an even higher level.

Considering only the electricity generation based on non-nuclear fuels, the share of CHP has been well over 50% between 1985 and 1995. In 1994 the share was 52%, which was the lowest figure during the past ten-year period. Because of anticipated increases in electricity demand, the share of fuel-based generation is expected to increase in the future. The condensing power plants are mainly based on pulverised coal combustion, with a total capacity of about 2.2 GW_e (end of 1995). The remaining public utility condensing plants are fuelled by peat (about 310 MW_e), natural gas (about 260 MW_e), and oil (about 260 MW_e).

Electricity generation in district CHP plants is mainly based on coal (43% in 1995), natural gas (30%), and peat (19%). Biomass accounts for about 4% of the electricity output. Within the industrial CHP generation, however, biomass is by far the largest primary energy source with a share of about 60% of the total electricity output in 1995. Natural gas accounts at present for about 15% of industrial electricity generation, while oil, coal, peat and process gases each have a share of 4–9%.

Solid fuel fired plants both in district and industrial CHP generation are increasingly based on fluidised bed combustion. New solid plants employ almost exclusively the fluidised bed (FBC) technology, and many older pulverised coal and peat fired plants have been converted into FBC plants. Industrial plants fuelled with waste liquors from pulping, however, need to employ a special recovery boiler technology due to the process chemical residues to be recovered from the fuel.

The amount of sulphur emissions from power production has steadily decreased since 1980 and the amount of nitrogen and particulate emissions since 1990. The total sulphur emissions attributable to electricity generation were in 1990 about 54 kt(SO_2) and nitrogen emissions were about 38 kt(NO_2) (Järvinen 1997). The corresponding amount of particulate emissions was approximately 10 kt in 1990. In these estimates the allocation of emissions from CHP to electricity and heat is based on efficiency factors that are somewhat lower for electricity than

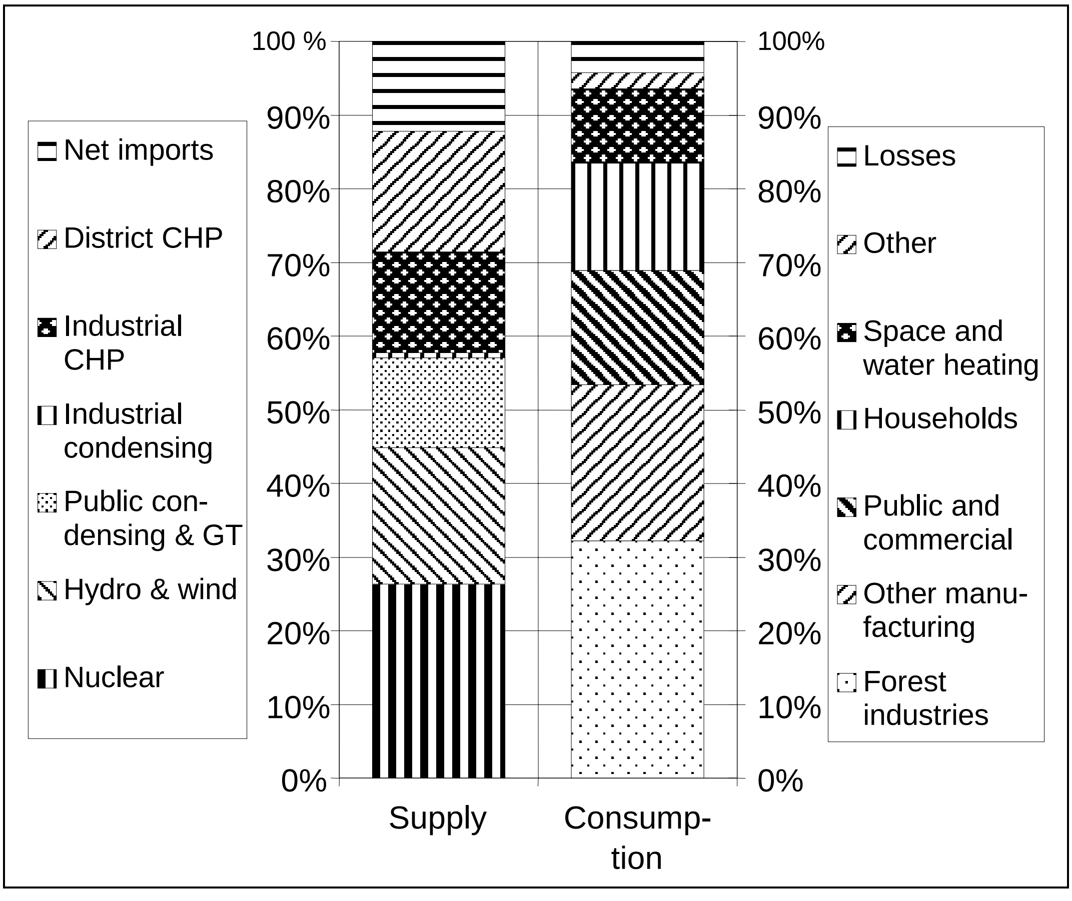


Figure 7.2 Structure of electricity supply and consumption in Finland in 1995.

for heat to reflect the trade-offs between CHP and pure heat generation. The recent emission estimates for the year 1995 are the following: sulphur 21 kt, nitrogen 25 kt (Heikkinen 1998) and particulates 6 kt (Lehtilä 1998).

The specific emissions have significantly decreased during the past decade as the total volume of power production has increased. The estimated average sulphur emissions from power production in Finland in 1990 were about 1.0 $g(SO_2)/kWh_e$, whereas the average among the present EU-15 countries was about 3.3 g/kWh (Järvinen 1997). Corresponding estimates of nitrogen oxide emissions are 0.73 $g(NO_2)/kWh_e$ for Finland and 1.2 g/kWh for EU-15. Within the next ten years the difference in specific emissions between Finland and the average over EU is expected to be preserved for sulphur emissions, but significantly decreased for nitrogen emissions. In 1995 the specific emissions of power production in Finland were 0.35 $g(SO_2)/kWh_e$ and 0.41 $g(NO_2)/kWh_e$ using the above emission estimates of Heikkinen (1998).

7.3. Aggregation methods

The damages or external costs of the whole Finnish electricity sector were aggregated from the fuel cycle results using two very simple methods. The first is a direct extrapolation of the National Implementation damage estimates (mECU/kWh) of each fuel cycle to the whole electricity generation (fired by the corresponding fuel) and could be called a simple bottom-up aggregation. In the second method the estimated total emissions of electricity generation in Finland were used as a basis for aggregation. Multiplying these by damage estimates (ECU/t pollution) and summing gives an estimate of the total damage of the emissions of electricity generation.

Strictly taken, aggregation by extrapolating simply the damages of an individual fuel cycle would require that the damages are linear functions of emissions, that the damages are not dependent on the location of the power plant and that all the other plants represent a similar technology with the same specific emissions. None of these assumptions are in reality valid but the aggregation results may still provide an approximate estimate of the total damages of pollutants. In the present ExternE framework the exposure-response functions for the most important non-synergetic health impacts and damages are linear without any threshold. The global warming damages are also assumed to be linearly dependent on the GHG emissions. However, the National Implementation studies have shown the sensitivity of the damage figures to the plant location (even within a single country) and stack height. The pollution abatement and firing technology determines the specific emissions, which vary from one power station to another.

The first aggregation method is clearly weaker because it is based solely on specific emissions (g/kWh) of the newer pollution abatement technology of the NI case plants whereas the average specific emissions in Finland are presumably higher. The second method is based on more realistic estimates of the total sulphur, nitrogen and particulate emissions in Finland given in section 7.2 above. However, the applied damage estimates of SO_2, NO_x and TSP

(ECU/t pollution) were only averages of the results of the three fuel cycles, which are also crude approximations and do not contain impacts on ecosystems.

The aggregation of total greenhouse gas emissions based on the individual fuel cycles is more realistic, and the impact of the GHGs is not dependent on the location of the power plant. On the other hand, knowledge of the global warming impacts is still very poor. Consequently the basic GHG damage estimates (ECU/t pollution), utilised in the ExternE Project, are very uncertain as discussed earlier.

7.4. Results

The summarised damage estimates of the NI fuel cycles and the aggregation results using the first method are shown in Table 7.2. The damages of nuclear and hydro power attributed to greenhouse gas, sulphur, nitrogen and particulate emissions are negligible. Coal, peat and biomass, on which the aggregation is based, corresponded in 1995 to approximately 74% of the non-nuclear fuel-based electricity generation. The other fuel-based electricity generation was left out from the aggregation.

The results of applying the second aggregation method to sulphur, nitrogen and particulate emissions are shown in Table 7.3. Here the basis is formed by the specific emissions (g/kWh) of the *whole* Finnish electricity generation sector in 1995, discussed in section 7.2 (Heikkinen 1988, Lehtilä 1998). From these figures the (average) specific emissions of *non-nuclear fuel-based* electricity generation in 1995 were calculated (first column of specific emissions in Table 7.3), which in the case of CHP are based mainly on the energy content of electricity.

If the exergy principle were applied, the specific emissions would be greater. The approximate specific emissions of this case are illustrated in the next column. The following assumptions were applied: 1) 50% of electricity was generated in CHP plants, 2) The burdens, impacts and damages allocated to electricity using the exergy principle are twofold those obtained using the energy principle. These exergy-based numbers are comparable with the specific emissions of the power generation stage of the NI fuel cycles in Table 7.1. The total emissions (in tonnes) in 1995 were calculated on the basis of these specific emissions and the known electricity production in 1995 given in Table 7.2.

The total damage estimates of Table 6.3 (last column) were then calculated by using average damages per tonne of pollutant estimated in the ExternE Finnish National Implementation Project. These total damage estimates for SO_2, NO_x and TSP are clearly greater than those obtained using the direct extrapolation of the NI fuel cycles in Table 7.2. In the simple bottom-up aggregation (based on the three case plants) the damage estimate in Table 7.2 is 64 MECU/ a whereas an estimate of 88 MECU/a is obtained in Table 7.3 (based on Finnish total emissions). Furthermore, only the power generation stage is included in Table 7.3.

The difference between the results is surprisingly small. It can be understood when comparing the specific emissions of the case plants (Table 7.1) and the approximate specific emissions of the whole Finnish electricity generation sector (Table 7.3), which are all of the same order of magnitude.

Table 7.2 Aggregation based on application of the ExternE NI results to the whole electricity generation in Finland.

ELECTRICITY GENERATION IN FINLAND			
Reference year	1995		
ELECTRICITY MIX	GWh /a	%	
Coal	9926	16%	
Nuclear	18130	30%	
Gas	6285	10%	
Oil	1438	2%	
Peat + Biomass	12032	20%	(from which biomass 10%, about 6060 GWh/a)
Hydro power	12790	21%	
Total	60600	*100%*	

ExternE NI, Finland	**Damages of fuel cycles**						**AGGREGATED DAMAGES OF ELECTRICITY SECTOR**				
SPECIFIC DAMAGES	**mECU/kWh**			**ECU/t**			**MECU/a**				
VOLY approach	SO_2	NO_x	TSP	SO_2	NO_x	TSP	SO_2	NO_x	TSP	Total	
Coal (Russian pop. incl)	1.23	0.83	0.26	1486	1310	1555	12	8	3	23	
Nuclear											negligible
Gas											no est.
Oil											no est.
Peat	1.98	1.01	0.19	1027	856	1344	12	6	1.1	19	
Biomass	0.68	2.33	0.56	1607	1388	2611	4	14	4	22	
Hydro power											negligible
										64	*Subtotal*
Global warming (mid 1% estimate)	CO_2	N_2O	CH_4	CO_2	N_2O	CH_4	CO_2	N_2O	CH_4		
Coal	36.56	0.24	2.76	46	14260	966	363	2	27	393	
Nuclear											negligible
Gas											no est.
Oil											no est.
Peat	43.64	0.50	-0.78	46	14260	966	261	3	-4.6	259	
Biomass	0.39	2.96	0.40	46	14260	966	2	18	2	23	
Hydro power											negligible
										738	***TOTAL***
										416	COAL
										278	PEAT
										44	BIOMASS

Remark: Damages from airborne emissions above only

Coal, peat and biomass:	
Portion of fuel-based electricity =	74%

Table 7.3 Emissions and estimated damages of the pollutants SO_2, NO_x, and TSP in Finnish electricity generation.

Non-nuclear fuel-based electricity generation in Finland	Specific emissions* Base yr 1995	Specific emissions** Base yr 1995	Estimated emissions in 1995**	Damage estimates of pollutants***	Total damages of power generation stage
	g / kWh$_e$	**g / kWh$_e$**	**t / yr**	**ECU / t**	**MECU**
SO_2	0.71	0.88	26,000	1395	37
NO_x	0.84	1.1	31,000	1208	38
TSP	0.20	0.25	7,500	1789	13
				TOTAL	**88**

* burdens allocated to electricity using the **energy** principle in CHP generation
** approximate figures if burdens were allocated to electricity using the **exergy** principle in CHP generation
*** approximate damage estimates calculated on the basis of a weighted average of the ExternE Finnish NI plants

The greenhouse gas emissions were aggregated by using the first method only. The GW damage estimate (1% discount rate, see Appendix V, Table 4: Illustrative restricted range, high estimate) dominates all the other damages caused by electricity generation in Finland.

7.4.1. Concluding remarks on aggregation

It should be noted that all the aggregation results regarding the damages caused by sulphur, nitrogen and particulate emissions are very uncertain as argued above. From the specific damages (ECU/t pollution) of the NI fuel cycles in Table 7.2 it can clearly be seen how they vary widely from plant to plant. It is difficult to make any generalisations from these monetary values to the whole electricity generation sector in Finland. The figures would probably be very specific for every power station and dependent both on the plant itself and its location. Because the damages caused to human health dominate in ExternE methodology, the distribution of population in a country is here decisive: the total damage is a function of the number of recipients. The sensitivity of the results to the plant location was not investigated in the Finnish study.

It should be noted that the technology of the NI plants does not represent the average technology of the fuel cycle in Finland. For example, most of the biomass utilised for energy is in the form of black liquor (in the pulp industry). The results for the ExternE plants are therefore not easily generalised to the whole biomass-based electricity and heat production. Furthermore, the fact that coal and peat are used both in condensing and in CHP plants makes the aggregation within these fuel cycles more uncertain.

The estimates presented above are meaningful only in the context of the ExternE methodology and the assumptions made in the case studies. The use of these estimates autonomously together with traditional financial data in decision-making is not justifiable.

8. Conclusions

In this study a large number of externalities for electricity generation were calculated on the basis of the methodology and theoretical work of the earlier ExternE Project (EC, 1995a-f) and the Core Project (EC, 1998). In this Finnish National Implementation three different fuel cycles were considered. The main tool for calculating the dispersion of airborne pollutants from power generation and the resulting environmental impacts and damages was the EcoSense model, which follows the so called impact-pathway approach of the ExternE Project.

The ExternE methodology is an attempt towards the integration of environmental impacts into energy policy. Here an additional object is the quantification of impacts in monetary terms so that the monetary results could be used in economic decision making according to the discipline of neo-classical environmental economics. The outcome of the monetary valuation of the impacts are the external or damage costs.

Different types of impacts could be identified with respect to their monetary valuation. Some identified impacts were not quantified at all in this National Implementation study, because earlier ExternE studies had shown them to be of minor importance (for the specific fuel cycle) or no quantification criteria were developed. One group are the impacts on natural ecosystems (through acid deposition), which were quantified in terms of critical load excess areas but for which no monetary valuation could be presented. Furthermore, the valued damages of most impacts were negligible compared to the price of electricity (and thus of little value in practical policy decisions).

In the Finnish study only human health related and global warming impacts appeared to be significant in monetary terms. Their external costs are so high that they could affect decisions in energy policy, although the health impacts were essentially lower than in some central European studies.

There are many uncertainties of diverse character in the results. In the result tables the statistical uncertainty of the exposure-response functions has been described by their uncertainty rating (A, B, C) given in Appendix VII. However, this is only one and certainly not the most important type of uncertainty involved.

The choice of the priority impact pathways considered in this study was mainly based on the experience gained in previous ExternE studies. It is possible that some important impact mechanisms were not considered at all. Furthermore the ‘atomistic’ approach with distinct impact pathways might lose some vital information on the value of synergistic effects, and thus the overall situation might be more than the aggregate of its parts. One undeniable fact is that understanding of the long-term impacts on natural ecosystems is rather poor. The same applies to the actual impacts of global warming.

One type of uncertainty is the validity and accuracy of the models for the chosen impact pathways or considered phenomena. For example, how accurate was the atmospheric dispersion model applied in the study?

The last part of the pathway is the monetary valuation of the impact, where the uncertainties are also related to the subjective factors of the valuation process.

There is a serious risk of misinterpretations when considering only the final results of the study — the total external costs or damages — without paying attention to the intermediate stages of the impact pathway. A qualitative impression of the externalities, i.e. external impacts before their monetary valuation, is also important as well as an understanding of the methodological limitations. The focus is too easily directed towards factors that are quantifiable and theoretically more susceptible, which can lead to bias in the interpretation of results. The uncertainty of scientific knowledge is a fact that should also be estimated in the decision-making procedures.

It has to be remembered that, even if the theoretical knowledge of impact mechanisms were developed, attempts towards monetary valuation and damage cost minimisation may not necessarily be the best approach to the problem. Some sustainability constraints may be a better policy instrument than pure social and environmental costs minimisation (Eyre, 1997).

It is also questionable whether the different externalities are commensurate in monetary terms. The decision problem concerning the externalities and energy policy is multidimensional, essentially value-laden and plural in character. For good reasons it can be claimed that no purely analytical procedure can provide a satisfactory solution for such a problem. In other words, there is no uniquely rational way to resolve contradictory perspectives or conflicts of interests. There can be no 'analytical fix' for the problems of environmental appraisal (Stirling, 1997). For example, the way in which human life is monetarily valued in the ExternE methodology determines to a large extent the level of external costs. It can be asked whether the valuation should be open to discussion and to a democratic process and not be chosen by scientists. The same certainly applies also to the choice of the prior impacts of the fuel cycles, which was made in the ExternE Project.

However, despite the uncertainties and limitations of the methodology, it can be an effective tool in the comparison of *similar kinds* of fuel cycles, to some extent for new power plant and pollution abatement technologies and different plant locations with each other. The relative differences may be more interesting than the absolute figures of external costs. The strength of the 'bottom-up' approach is that the analysis is as much as possible case-specific, taking into account most of of the concrete details of the fuel cycle under consideration. Impact classes, which were omitted in the cost analysis, should be valuated by other means.

Much more uncertain is the generalisation or aggregation of the results to the whole electricity generation sector, to the total external costs due to its airborne emissions. The three fuel cycles assessed in this National Implementation do not represent a mean of the whole non-nuclear fuel-based electricity generation in Finland but rather the newest technology, according to the objectives of the project. Consequently, the specific emissions of these individual fuel cycles and their estimated damages (mECU/kWh) are not a good basis for aggregation. Furthermore the damage estimates per tonne of air pollution (ECU/t) of this National Implementation Project appear to be very case specific, dependent on the location and stack height of the plant. A 'top-down' approach would probably be a more fruitful basis for estimating the external costs of the total electricity generation sector.

To conclude, the ExternE methodology is directly applicable to certain case-specific comparisons of alternative electricity generating options. It also provides insight concerning the relative importance of various environmental issues in more general settings. However, the inherent limitations of the approach and the currently incomplete knowledge of certain damage chains limit its usefulness in more general policy issues or generic comparison of generating options.

9. References

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APPENDICES

I. THE ECOSENSE MODEL

I.1. Introduction

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

Taking into account this background, the objectives for the development of the EcoSense model were:

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

As health and environmental impact assessment is a field of large uncertainties and incomplete, but rapidly growing understanding of the physical, chemical and biological mechanisms of action, it was a crucial requirement for the development of the EcoSense system to allow an easy integration of new scientific findings into the system. As a consequence, all the calculation modules (except for the ISC-model, see below) are designed in a way that they are a *model-interpreter* rather than a *model*. Model specifications like e. g. chemical equations, dose-response functions or monetary values are stored in the database and can be modified by the user. This concept allows an easy modification of model parameters, and at the same time the model does not necessarily appear as a black box, as the user can trace back what the system is actually doing.

I.2. Scope of the EcoSense model

EcoSense was developed to support the assessment of priority impacts resulting from the exposure to airborne pollutants, namely impacts on health, crops, building materials, forests, and ecosystems. Although global warming is certainly among the priority impacts related to air pollution, 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.

I.3. The EcoSense Modules

Figure I-1 shows the modular structure of the EcoSense model. All data - input data, intermediate and final results - are stored in a relational database system. The two air quality models integrated in EcoSense are stand-alone models, which are linked to the system by pre- and postprocessors. There are individual executable programs for each of the impact pathways, which make use of common libraries. The following sections give a more detailed description of the different EcoSense modules.

I.3.1. The EcoSense database

I.3.1.1 *Reference Technology Database*

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

I.3.1.2 Reference Environment Database

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

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

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

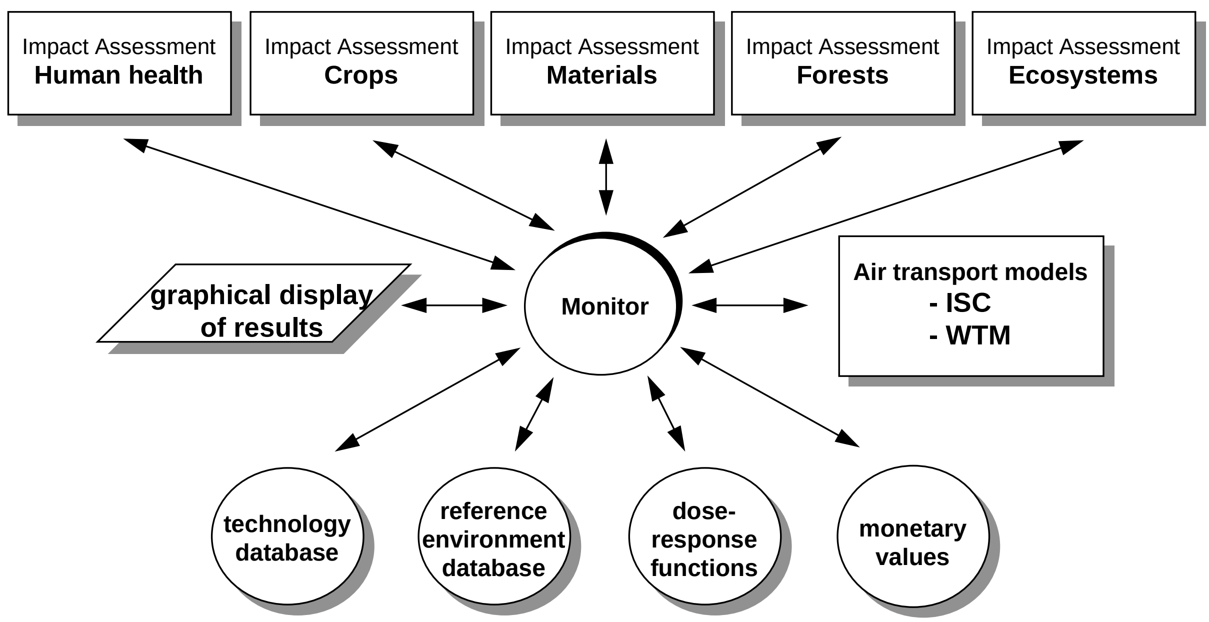

Figure I-1 *Structure of the EcoSense model.*

I.3.1.3 Exposure-Response Functions

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

I.3.1.4 Monetary Values

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

I.3.2. Air Quality Models

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

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

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

I.3.3. Impact Assessment Modules

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

I.3.4. Presentation of Results

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

I.4. The air quality models integrated in EcoSense

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

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

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

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

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

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

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

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

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

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

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

where:

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

- The mixing height is terrain following.

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

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

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

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

The Windrose Trajectory Model (WTM) used in EcoSense to estimate the concentration and deposition of acid species on a regional scale was originally developed at Harwell Laboratory by Derwent and Nodop (1986) for atmospheric nitrogen species, and extended to include sulphur species by Derwent, Dollard and Metcalfe (1988). The model is a receptor-orientated Lagrangian plume model employing an air parcel with a constant mixing height of 800 m moving with a representative wind speed. The results are obtained at each receptor point by considering the arrival of 24 trajectories weighted by the frequency of the wind in each 15° sector. The trajectory paths are assumed to be along straight lines and are started at 96 hours from the receptor point. The chemical scheme of the model is shown in Figure I-2.

In EcoSense, the model is implemented by means of

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

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

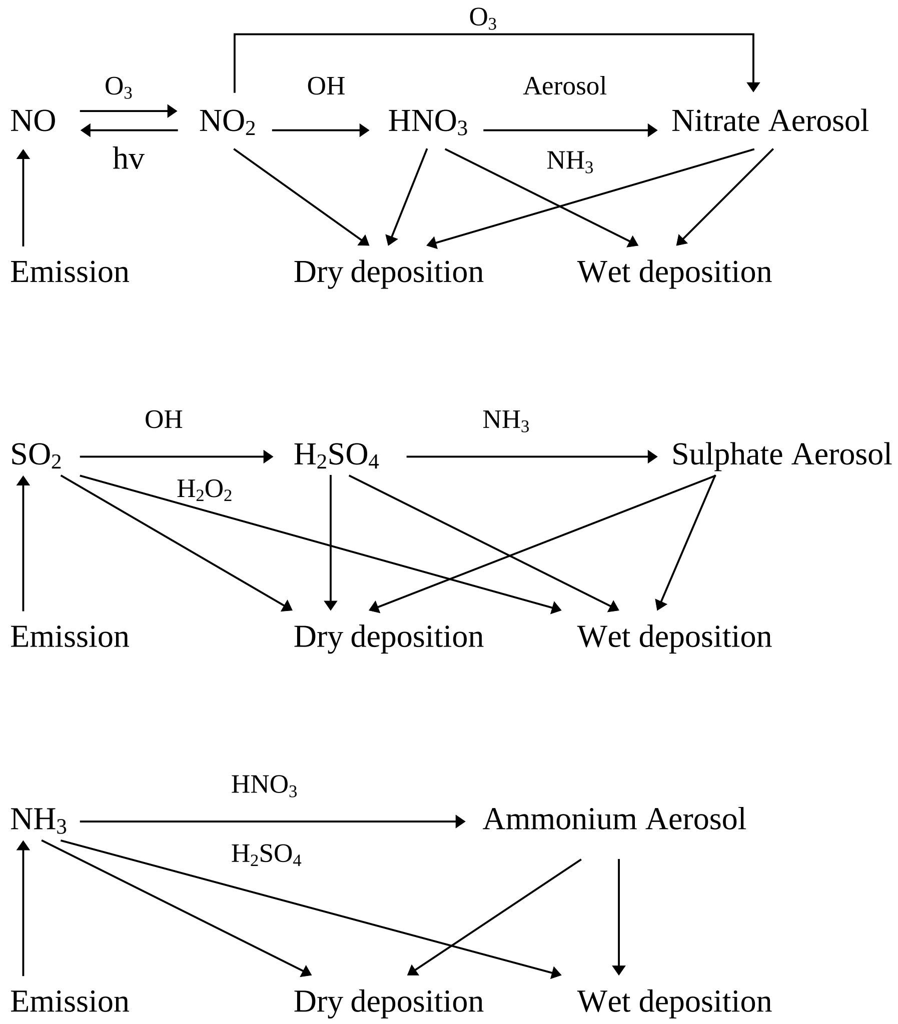

Figure I-2 *Chemical Scheme in WTM, adopted from Derwent et al. (1993)*

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II. HEALTH EFFECTS

II.1. Introduction

Five types of health effect have been dealt with in the present study;

1. Non-carcinogenic effects of air pollutants
2. Carcinogenic effects of radionuclide emissions
3. Carcinogenic effects of dioxins and trace metals
4. Occupational health issues (disease and accidents)
5. Accidents affecting members of the public

Each of these is discussed briefly below, followed by a review of valuation issues for health effects. A more complete description of the assumptions made is given in the ExternE methodology report (European Commission, 1998), and for carcinogenic effects of radionuclides in the earlier report on the nuclear fuel cycle (European Commission, 1995e). It has to be noted that, since the results of ExternE 1995 (European Commission, 1995a-f) were published, a lot of new information has become available, changing the quantification and valuation of some health impacts significantly.

II.2. Non-Carcinogenic Effects of Air Pollutants

II.2.1. Introduction

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

NO_x	SO_2	NH_3	CO
ozone	nitrate aerosol	sulphate aerosol	PM_x

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

II.2.2. Epidemiological evidence

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

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

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

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

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

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

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

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

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

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

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

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

Additional suggested sensitivity analyses:

(1) Try omitting SO_2 impacts for acute mortality and respiratory hospital admissions;

(2) Treat all particles as PM_{10} or $PM_{2.5}$;

(3) Try omitting all RADs and MRADs;

(4) Scale down by 2 the E-R functions for chronic mortality by Pope *et al*.

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

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

a) What ambient air pollutants have been shown as *associated* with adverse health effects (acute or chronic), and for what specific endpoints;

b) Which of these associations may reasonably be interpreted as *causal*; it is important in assessing the effect of *incremental* pollution in ExternE to quantify *causal* relationships, and not just epidemiological associations).

c) What studies provide a basis for a good set of E-R functions, for quantifying the public health effects of incremental air pollution; and

d) How if at all should the E-R functions from individual studies be adapted for use in ExternE.

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

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

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

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

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

In ExternE 1995, we concluded that the evidence for SO_2 damaging health was too weak for functions to be recommended. However, in the APHEA studies, the size of the apparent SO_2 effect did not depend on the background concentrations of ambient particles. In the context of the evidence as a whole, including this result, it is recommended that the functions for SO_2 are used in the main ExternE implementations now; and that the estimated impacts are added to the effects of particles and of ozone.

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

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

b) the functions for CO and acute mortality are not used.

Sensitivity analyses should consider including both, or omitting both.

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

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

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

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

II.3. Carcinogenic Effects of Radionuclide Emissions

II.3.1. Introduction

A brief explanation of the terminology specific to the nuclear fuel cycle assessment is presented in Box 1. Unlike the macropollutants described in the previous section, analysis of the effects of emissions of radionuclides is not carried out using the EcoSense model (it was not felt necessary, or practicable, to include every impact pathway for fuel chain analysis within EcoSense). In view of this it is necessary to give additional details of the methodology for assessment of the damages resulting from radionuclide emissions, compared to the information given in the other sections in this Appendix. The details given relate specifically to the French implementation of the nuclear fuel cycle (European Commission, 1995e). For the implementation in the present phase of the study, a more simplified approach has been adopted by some teams that extrapolates from the French results.

Box 1 Definitions

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

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

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

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

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

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

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

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

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

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

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

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

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

The evaluation of severe reactor accidents are treated separately due to their probabilistic nature and the need to use a different type of atmospheric dispersion model (European Commission, 1998), though the principles for quantification of impacts remain the same as described here. Differences arise at the valuation stage.

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

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

II.3.2. Boundaries of the Assessment

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

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

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

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

II.3.3. Impacts of atmospheric releases of radionuclides

The most important impact pathways for public health resulting from atmospheric releases are:

- inhalation and external exposure due to immersion from the radionuclides in the air,
- external exposure from ground deposition, and
- ingestion of contaminated food resulting from ground deposition.

These pathways are illustrated in Figure 2.

II.3.3.1 Dispersion

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

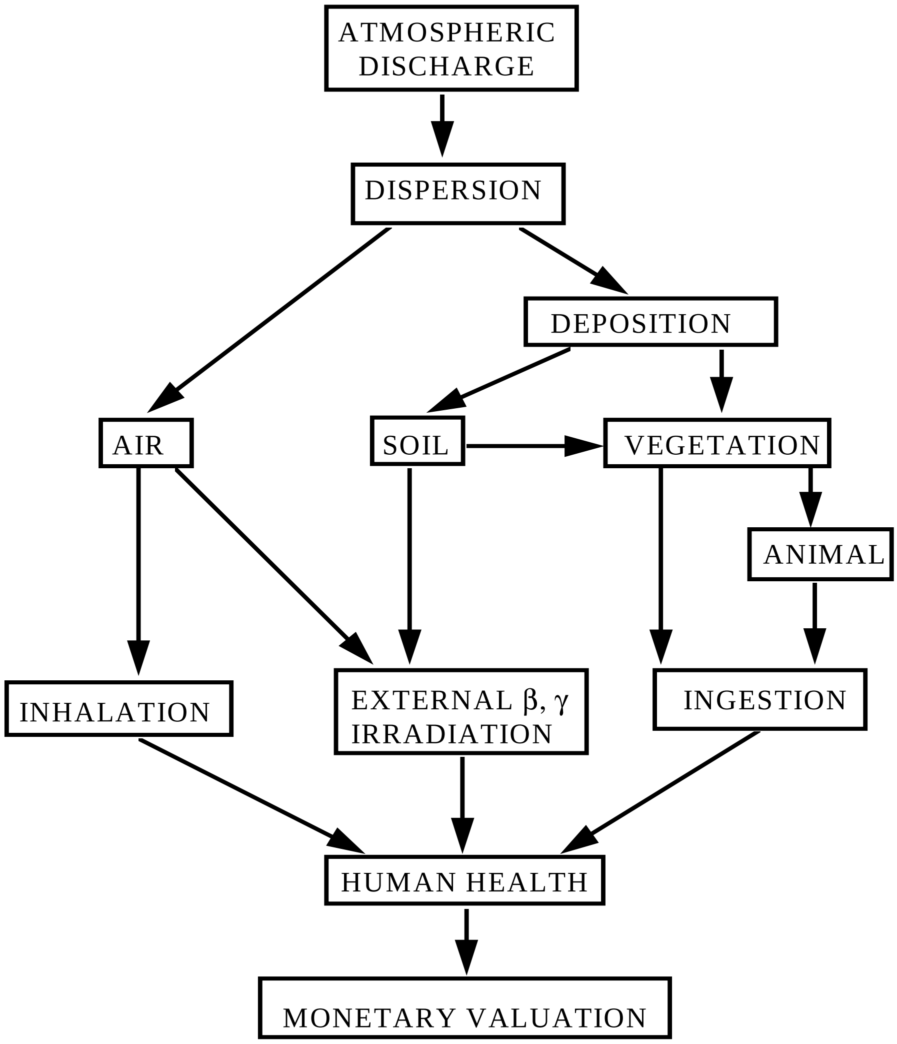

***Figure II-1** Impact pathway for an atmospheric release of radionuclides into the terrestrial environment.*

II.3.3.2 Exposure

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

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

The human consumption pathway via agricultural products arises from direct deposition on the vegetation and migration of the radionuclides through the roots via the soil. Again, depending on the environmental and physical half-lives of each radionuclide, the time scale of importance varies but it is considered that 100,000 years should be sufficient.

A detailed environmental pathway model has not been used here. The environmental transfer factors between deposition and food concentration in different food categories, integrated over different time periods, assuming generalised European agricultural conditions was obtained from the NRPB agricultural pathway model FARMLAND. A constant annual deposition rate is assumed and the variation in the seasons of the year are not taken into account. The agricultural products are grouped, for this generalised methodology, as milk, beef, sheep, green vegetables, root vegetables and grains. Examples of the transfer factors used for a few radionuclides are given in **Table II-3**.

Cultivated vegetation is either consumed directly by people or by the animals which ultimately provide milk and meat to the population. The exposures received by the population are calculated taking into consideration food preparation techniques and delay time between harvest and consumption to account from some loss of radioactivity. An average food consumption rate data (illustrated by the French data shown in **Table II-4**) and population size is used for calculating the amount of food that is consumed in the local, regional and global population. The collective doses are calculated assuming that the food will be consumed locally but if there is an excess of agricultural production it will pass to the regional population next, and afterwards to the global population group. In this way the dose due to the total food supply produced within the 1000 km area included in the atmospheric dispersion assessment is taken into account.

II.3.3.3 Dose Assessment

It is possible to report a calculated dose by radionuclide, type of exposure and organ of the body, but for the purpose of estimating a population risk, a whole body effective collective dose was calculated taking into account these factors. A few examples of the dose conversion factors used in the evaluation are presented in **Table II-5**.

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

Table II-3 *Food transfer coefficients, integrated over different time periods, for food products (in Bq/kg per Bq/m^2/s of deposition)*

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

Table II-4 *Average consumption rates for an average French adult.*

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

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

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

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

Table II-5 *Dose conversion factors for exposure by ingestion and inhalation of radionuclides (Sv/Bq).*

Radionuclide	Half-life	Type of release	Type of exposure	Dose conversion factor (Sv/Bq)
H-3	12.3 y	Liquid, gaseous	Ingestion	1.80 E-11
			Inhalation	1.73 E-11
C-14	5710 y	Liquid, gaseous	Ingestion	5.60 E-10
			Inhalation	5.60 E-10
I-129	1.6 E7 y	Gaseous	Ingestion	1.10 E-07
			Inhalation	6.70 E-08
I-131	8.1 d	Liquid, gaseous	Ingestion	2.20 E-08
			Inhalation	1.30 E-08
Cs-134	2.1 y	Liquid, gaseous	Ingestion	1.90 E-08
			Inhalation	1.20 E-08
Cs-137	30 y	Liquid, gaseous	Ingestion	1.30 E-08
			Inhalation	8.50 E-09
U-234	2.5 E5 y	Liquid, gaseous	Ingestion	3.90 E-08
			Inhalation	2.00 E-06
U-235	7.1E8 y	Liquid, gaseous	Ingestion	3.70 E-08
			Inhalation	1.80 E-06
U-238	4.5 E9 y	Liquid, gaseous	Ingestion	3.60 E-08
			Inhalation	1.90 E-06
Pu-238	86.4 y	Liquid, gaseous	Ingestion	2.60 E-07
			Inhalation	6.20 E-05
Pu-239	2.4 E4 y	Liquid, gaseous	Ingestion	2.80 E-07
			Inhalation	6.80 E-05

II.3.3.4 Time Distribution of the Expected Occurrence of Health Effects

The use of the dose response functions provides the estimate of the total number of health effects expected; however, the details on the expected time of occurrence of these effects 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.

II.3.4. Impacts of liquid releases of radionuclides

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

II.3.4.1 River

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

The deposition of the radionuclides in the irrigation water to the surface of the soil and transfer to agricultural produce is assumed to be the same as for atmospheric deposition.

The ingestion pathway doses are calculated in the same way as described above for the atmospheric pathway. For aquatic releases, it is difficult to calculate independent local and regional collective doses without creating extremely simplified and probably incorrect food distribution scenarios. Therefore, the local and regional collective doses are reported in the regional category. The estimation of health effects also follows the same methodology as described in the section above.

II.3.4.2 Sea

To evaluate the collective dose due to consumption of seafood and marine fish, a compartment model which divides the northern European waters into 34 sections was used for the original French implementation. This model takes into account volume interchanges between compartments, sedimentation, and the radionuclide transfer factors between the water, sediment, fish, molluscs, crustaceans, and algae, and the tons of fish, molluscs, crustaceans and algae harvested for consumption from each compartment. For the regional collective dose, it is assumed that the 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.

II.3.5. Impacts of releases of radionuclides from radioactive waste disposal sites

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

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

II.3.6. Impacts of accidental atmospheric releases of radionuclides

The methodology used to evaluate impacts due to accidental releases is risk-based expected damages. Risk is defined as the summation of the probability of the occurrence of a scenario

(P_i) leading to an accident multiplied by the consequences resulting from that accident (C_i) over all possible scenarios. This can be simply represented by the following equation:

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

II.3.6.1 Transportation accidents

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

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

II.3.6.2 Severe Reactor Accidents

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

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

II.3.7. Occupational impacts from exposure to radiation

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

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

II.3.8. Impacts of transportation on human health

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

II.4. Carcinogenic Effects of Dioxins and Trace Metals

The basic impact assessment approach used in ExternE for macropollutants (see above) is still valid for the micropollutants - after all it simply seeks to quantify the pathway from emission to impact and monetary damage. However, the step in which incremental exposure of the stock at risk is quantified requires elaboration to account for both direct and indirect exposure. The range of possible exposure pathways is shown in **Table II-6**.

Table II-6 Exposure pathways for persistent micropollutants.

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

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

II.4.1. Dioxins and Dibenzofurans

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

II.4.1.1 Threshold levels

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

The position of the World Health Organisation (French Academy of Sciences, 1995) is that the tolerable daily intake (TDI) is 10 $pg/kg_{bw}\cdot day$ (10^{-12}g per kg body weight per day). The TDI represents an average lifetime dose, below which damage is considered unlikely. Calculation of the TDI involves the use of safety factors, which is illustrated in **Table II-7**. Safety factors reflect the uncertainty involved in extrapolating data between species and also the perceived severity of the effect.

Table II-7 *Use of safety factors in setting guideline intake levels (DoE, 1989).*

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

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

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

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

II.4.1.2 Pathway analysis for dioxins

Considerable debate has surrounded the calculation of human exposure to dioxins from incineration. Whilst early studies concentrated on the direct (inhalation) exposure route, more recent analyses have modelled the transfer of the contaminants from the incinerator to the exposed population via most, or all of the routes shown in **Table II-6**.

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

The analysis covered background exposure and incremental exposure due to the incinerator. This allowed assessment of the relative importance of the different sources of the total dose, and, since the study used the WHO threshold value to assess health effects, an assessment of the net risk to the population from dioxin intake. **Table II-8** summarises the results for the plant emitting the highest levels of dioxins considered by HMIP.

Table II-8 *Summary of mean dioxin intakes for an adult HMEI* living close to an incinerator sited in urban and rural locations.*

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

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

It can be seen from **Table II-8** that even in the worst case considered, of an urban HMEI living near the largest plant emission considered by the study, the total intake does not approach the WHO threshold level. However, recent studies have suggested that the dioxin intake of an average breast-fed baby could be as high as 110 pg/kg/day at two months, falling to 25 pg/kg/day at ten months. Results from the HMIP study are not directly comparable (being averaged over a longer period), but also suggest exposure above the WHO recommended TDI of 10 pg/kg/day. However, we adopt the position of recent reviews (DoH, 1995), that when averaged over a lifetime, the cumulative effect of increased dioxin intake during breast feeding is not significant. The HMIP study concluded that emissions of dioxins from municipal waste incinerators operating to EU legislative standards do not pose a health risk to individuals, irrespective of the location and size of the incinerator or the exposed population.

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

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

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

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

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

Where

C: concentration (mg/m^3)

IR: inhalation rate (m^3/hour)

ET: exposure time (hours/day)

EF: exposure frequency (hours/year)

ED: exposure duration (years)

BW: body weight (kg)

AT: averaging time

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

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

From this equation, and the expression of the I-TEQ per unit body weight it is apparent that body weight needs to be accounted for. Assumed values for body weight and IR*ET are shown in **Table II-9**.

Table II-9 *Assumptions for calculating inhaled dose.*

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

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

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

The relation between dose and concentration then is

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

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

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

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

No-threshold assumption:

Unit risk factor from

LAI 1.4 per 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

II.4.2. Impact Assessment for Heavy Metals

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

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

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

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

The general form of the exposure-response function is as follows for all cancer effects;

No. of additional cancers = Δ Concentration [$\mu g/m^3$] * unit risk factor * Population /70

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

II.4.2.1 Cadmium

Cancer

Unit risk factors from

ATSDR (1989)	0.0018 per $\mu g/m^3$
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.

II.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.

II.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.

II.4.2.4 Chromium

Cancer

Unit risk factor from

WHO (1987) 0.04 per µg/m^3

Non-carcinogenic effects

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

II.4.2.5 Nickel

Cancer

Unit risk factors from

WHO (1987) 0.0004 per µg/m^3

US-EPA (1996) 0.004 per µg/m^3

Non-carcinogenic effects

Threshold:

ATSDR (1996) 0.02 mg/(kg$_{BW}$·d)

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

Background:

WHO (1987)

rural areas: 0.1 - 0.7 ng/m^3

urban areas: 3 - 100 ng/m^3

industrial areas: 8 - 200 ng/m^3

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

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

II.5.1. Sources of data

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

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

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

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

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

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

The ExternE methodology aims to quantify all occupational health impacts, including those outside Europe, e.g. linked to the mining and treatment of imported fuels. Within the present phase of the study an assessment of damages outside Europe became necessary, partly because of changing market conditions (the UK for example is no longer entirely dependent on domestic coal mines), and partly because of the inclusion of cases where the countries concerned do not have an indigenous supply of fuel. Occupational health data have therefore been collected for as many countries and fuel chains as possible in the present phase of the study.

II.6. Accidents Affecting Members of the Public

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

II.7. Valuation

II.7.1. Introduction

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

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

- Mortality from fuel chain accidents that affect the public
- Injury from fuel chain accidents that affect the public

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

II.7.2. Mortality

The value of statistical life (VSL), essentially a measure of WTP for reducing the risk of premature death, is an important parameter for all fuel chains. A major review of studies from Europe and the US , covering three valuation methods (wage risk, contingent valuation and consumer market surveys) is described in earlier work conducted by the ExternE programme (European Commission, 1995b). The value derived for the VSL was 2.6 MECU. This value has been adjusted to 3.1 MECU, to bring it into line with January 1995 prices, as has been done for all valuations.

However, in earlier phases of the project a number of questions were raised regarding the use of the VSL for every case of mortality considered. These originally related to the fact that many people whose deaths were linked to air pollution were suspected of having only a short life expectancy even in the absence of air pollution. Was it logical to ascribe the same value to someone with a day to live as someone with tens of years of remaining life expectancy? Furthermore, is it logical to ascribe the full VSL to cases where air pollution is only one factor of perhaps several that determines the time of death, with air pollution playing perhaps only a minor role in the timing of mortality?. In view of this the project team explored valuation on the basis of life years lost. For quantification of the value of a life year (YOLL) it was necessary to adapt the estimate of the VSL. This is not ideal by any means (to derive a robust estimate primary research is required), but it does provide a first estimate for the YOLL.

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

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

The YOLL approach is particularly recommended for deaths arising from illnesses linked to exposure to air pollution. The value will depend on a number of factors, such as how long it takes for the exposure to result in the illness and how long a survival period the individual has after contracting the disease. On the basis of the best data available at the time, two sets of values have been estimated for impacts caused by fine particles: one for acute mortality and for chronic mortality (**Table II-10**). Both sets vary with discount rate.

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

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

II.7.3. Morbidity

Updated values for morbidity effects are given in **Table II-11**. Compared to the earlier report (European Commission, 1995b) most differences reflect an inflation factor, but some new effects are included, notably chronic bronchitis, chronic asthma, and change in prevalence of cough in children.

Table II-11 *Updated values in ECU for morbidity impacts.*

Endpoint	New Value	Estimation Method and Comments
Acute Morbidity		
Restricted Activity Day (RAD)	75	CVM in US estimating WTP. Inflation adjustment made.
Symptom Day (SD) and Minor Restricted Activity Day	7.5	CVM in US estimating WTP. Account has been taken of Navrud's study, and inflation.
Chest Discomfort Day or Acute Effect in Asthmatics (Wheeze)	7.5	CVM in US estimating WTP. Same value applies to children and adults. Inflation adjustment made.
Emergency Room Visits (ERV)	223	CVM in US estimating WTP. Inflation adjustment made.
Respiratory Hospital Admissions (RHA)	7,870	CVM in US estimating WTP. Inflation adjustment made.
Cardiovascular Hospital Admissions	7,870	As above. Inflation adjustment made.
Acute Asthma Attack	37	COI (adjusted to allow for difference between COI and WTP). Applies to both children and adults. Inflation adjustment made.
Chronic Morbidity		

Endpoint	New Value	Estimation Method and Comments
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.

II.7.4. Injuries

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

Table II-12 *Valuation data for injuries.*

ENDPOINT	VALUE (ECU, 1995)	ESTIMATION METHOD AND COMMENTS
Occupational Injuries (minor)	78	French compensation payments, increased for inflation.
Occupational Injuries (major)	22,600	French compensation payments increased for inflation.
Workers & Public Accidents (minor)	6,970	TRL (1995). New estimates.
Workers & Public Accidents (major)	95,000	TRL (1995). New estimates.

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

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

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III. AIR POLLUTION EFFECTS ON MATERIALS

III.1. Introduction

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

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

III.2. Stock at Risk Data

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

Eastern Europe:

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

Scandinavia:

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

UK, Ireland:

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

Greece:

NTUA (1997)

Germany, other Western Europe:

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

III.3. Meteorological, Atmospheric and Background Pollution data

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

For the UK:

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

For Germany;

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

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

III.4. Identification of Dose-Response Functions

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

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

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

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

ER	=	erosion rate (µm/year)
P	=	precipitation rate (m/year)
SO_2	=	sulphur dioxide concentration (µg/m^3)
O_3	=	ozone concentration (µg/m^3)
H^+	=	acidity (meq/m^2/year)
R_H	=	average relative humidity, %
f_1	=	$1\text{-}\exp[-0.121.R_H/(100\text{-}R_H)]$
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:

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

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

III.4.1. Natural stone

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

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

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

ICP - unsheltered limestone (4 years):

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

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

ICP - unsheltered sandstone (4 years):

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

ICP - sheltered limestone (4 years):

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

ICP - sheltered sandstone (4 years):

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

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

III.4.2. Brickwork, mortar and rendering

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

III.4.3. Concrete

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

III.4.4. Paint and polymeric materials

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

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

Haynie - carbonate paint:

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

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.

III.4.5. Metals

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

Although dose-response functions exist for many metals, this analysis is confined to those for which good inventory data exists; steel, galvanised steel/zinc and aluminium. Other metals could be important if the material inventories used were more extensive, quantifying for example copper used in historic monuments. Steel is typically coated with paint when not galvanised (see section 4.5.1 of this Appendix). The stock of steel in our inventories has therefore been transferred to the paint stock at risk.

III.4.5.1 Zinc and galvanised steel

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

Lipfert - unsheltered zinc (annual loss):

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

Butlin - unsheltered zinc (one year):

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

ICP - unsheltered zinc (4 years):

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

ICP - sheltered zinc (4 years):

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

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

III.4.5.2 Aluminium

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

Lipfert - aluminium (annual loss):

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

ICP - unsheltered aluminium (4 year):

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

III.5. Calculation of repair frequency

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

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

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

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

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

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

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

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

Material	**ECU/m^2**
Zinc	25
Galvanised steel	30
Natural stone	280
Rendering, mortar	30
Paint	13

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

III.7. Estimation of soiling costs

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

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

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

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

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

III.8. Uncertainties

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

- Improvement of inventories, in particular; the inclusion of country specific data for all parts of Europe; disaggregation of the inventory for paint to describe the type of paint in use; disaggregation of the inventory for galvanised steel to reflect different uses; disaggregation of calcareous stone into sandstone, limestone, etc. In addition, alternatives to the use of population data for extrapolation of building inventories should be investigated.

- Further development of dose-response functions, particularly for paints, mortar, cement render, and of later, more severe damage mechanisms on stone;

- Assessment of exposure dynamics of surfaces of differing aspect (horizontal, sloping or vertical), and identification of the extent to which different materials can be considered to be sheltered;

- Definition of service lifetimes for stone, concrete and galvanised steel;

- Integration of better information on repair techniques;

- Data on cleaning costs across Europe;

- Improvement of awareness of human behaviour with respect to buildings maintenance;

- The extension of the methodology for O_3 effects, including development of dose-response functions and models atmospheric transport and chemistry.

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

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

- Effects on historic buildings and monuments with "non-utilitarian" benefits;

- Damage to utilitarian structures that were not included in the inventory;

- Damage to paint work through mechanisms other than acid erosion;

- Damage to reinforcing steel in concrete;

- Synergies between different pollutants;

- Impacts of emissions from within Europe on buildings outside Europe;

- Impacts from ozone;

- Macroeconomic effects.

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IV. ANALYSIS OF ECOLOGICAL EFFECTS

IV.1. Introduction

Fuel chain activities are capable of affecting ecosystems in a variety of ways. This Appendix deals specifically with effects of air pollution on crop yield, on forest health and productivity, and effects of nitrogen on critical loads exceedence. It is based on an earlier review under the ExternE Project (European Commission, 1995a) which has been updated by Jones *et al* (1997, for inclusion in the updated ExternE Methodology Report, European Commission, 1998a).

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

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

Approaches for dealing with local impacts on ecology, for example, effects of transmission lines on bird populations, were discussed in the earlier ExternE report on the hydro fuel cycle (European Commission, 1995b). 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.

IV.2. Air Pollution Effects on Crops

IV.2.1. SO_2 Effects

A limited number of exposure-response functions dealing with direct effects of SO_2 on crops are available. Baker *et al* (1986) produce the following function from work on winter barley;

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

Where SO_2 = annual mean SO_2 concentration, ppb.

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

1. 0% yield reduction at 0 ppb and also at the value predicted by equation (1);

2. Maximum yield increase at an SO_2 concentration midway between the 2 values for which 0% yield effect is predicted from (1);

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

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

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

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

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

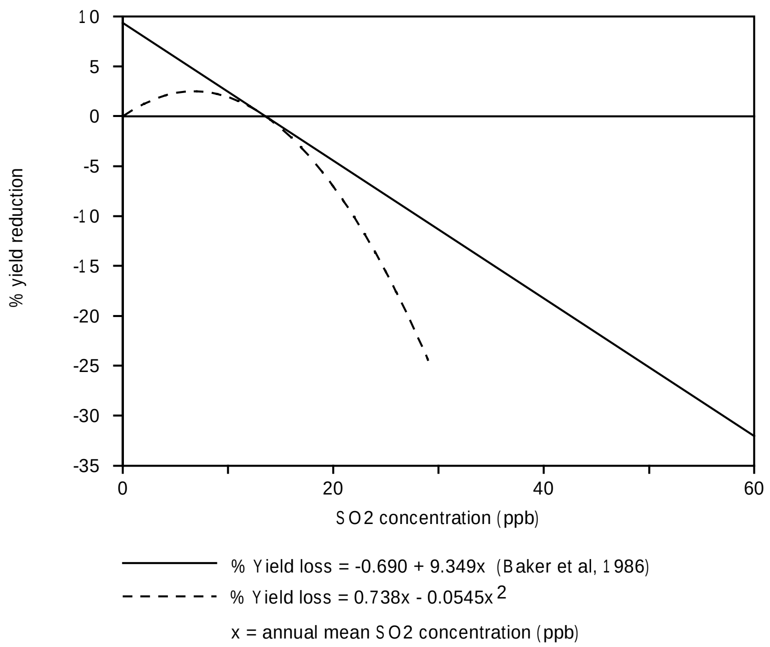

Figure 1. *Extrapolation of exposure-response functions below the lowest exposure level used experimentally.*

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

Weigel *et al* (1990) studied several crop cultivars common in Germany. Two spring barley cultivars ('Arena', 'Hockey'), two bean cultivars ('Rintintin', 'Rosisty') and one rape cultivar ('Callypso') were exposed to five different SO_2 levels between 7 and 202 µg m^{-3} (2.5 - 70 ppb) in open-top chambers. Exposure periods ranged from 49 to 96 days. 8 h/daily mean O_3-concentration ranged between 14 and 19 µg m^{-3}. Daily means of NO_2 and NO concentrations were generally lower than 10 µg m^{-3}. Yield increases appeared in all SO_2 treatments for the rape cultivar compared with controls whereas beans and barley were quite SO_2 sensitive. The probable cause of the positive response of rape was the high sulphur demand of this species (McGrath and Withers, 1996). Data for barley were taken from this paper and used to calculate the following relationship (SO_2 in µg m^{-3}):

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

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

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

The background mean SO_2 concentrations that provided the control levels in this study were low (7 - 9 µg 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 de-toxify any SO_2 absorbed.

IV.2.2. O_3 Effects

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

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

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

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

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 1.

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

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

relative to current agricultural production.

Exposure-response functions describing the action of ozone on crops have recently been developed using European data (Skärby *et al*, 1993). However, only 3 crops were covered, spring wheat, oats and barley, the last 2 of which were found to be insensitive to O_3. An expert panel on crop damage convened under the ExternE Project concluded that rye was also unlikely to be sensitive to O_3.

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

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

Spinach*	7 hr/day	135	2.08	Fuhrer *et al*, 1989
Lettuce*	7 hr/day	122	8.837	Fuhrer *et al*, 1989
Tomato*	7 hr/day	142	2.369	Fuhrer *et al*, 1989

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

The following function was derived for sensitive crops;

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

Where Y_{rel} = relative yield

x_8 = average daily peak 8 hour concentration.

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

The functions shown here refer to peak concentrations during 7, 8 or 12 hr periods. Ozone related crop damages were assessed against 6 hour peak values reported by Simpson (1992; 1993), generated from the EMEP model (Eliasson and Saltbones, 1983; Simpson, 1992). This model extends to the whole of Europe with a resolution of 150 km by 150 km. In addition the Harwell Global Ozone model (Hough, 1989; 1991) was also used, extending the zone of analysis to the whole of the Northern Hemisphere, though with greater uncertainty compared to the European analysis. Based on the these model results and the listed ozone crop functions an ozone crop damage factor of 490 ECU per tonne NO_x emitted in Europe has been derived.

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

	Average	**Standard Deviation**
$\frac{1}{y}$ dy/dConc from d-r function	-0.0058	0.0033
(y - 1)/Conc straight line	-0.0025	0.0014

IV.2.3. Acidification of Agricultural Soils

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

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

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

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

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

100 kg $CaCO_3$ is sufficient to neutralise 2 kg H^+. Accordingly the total acidifying pollution input on soils which require lime was multiplied by 50 to give the amount of lime which required to neutralise it. Further details were given in European Commission (1995a).

IV.2.4. Fertilisational Effects of Nitrogen Deposition

Nitrogen is an essential plant nutrient, applied by farmers in large quantity to their crops. The deposition of oxidised nitrogen to agricultural soils is thus beneficial (assuming that the dosage of any fertiliser applied by a farmer is not excessive). The analysis is conducted in the same way as assessment of effects of acidic deposition. The benefit is calculated directly from the cost of nitrate fertiliser, ECU 430/tonne of nitrogen (note: not per tonne of nitrate) (Nix, 1990). Given that additional inputs will still be needed under current conditions to meet crop N requirements there is a negligible saving in the time required for fertiliser application (if any).

IV.3. Modelling Air Pollution Damage to Forests

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

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

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

In the NAPAP review Kiester (1991) concluded that;

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

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

In the absence of directly applicable models for assessment of the effects of fuel cycle emissions on forests, further work, some of it conducted as part of the ExternE Project (European Commission, 1995a, Chapter 9), has sought to develop novel approaches to the assessment of forest damage in the last few years. The 1995 ExternE report paid particular attention to the studies by Sverdrup and Warfvinge (1993) and Kuylenstierna and Chadwick (1994), and functions developed by FBWL (1989) and Kley *et al* (1990). However, although we regard these approaches as worthy of further consideration, the results that they provide are too uncertain for application at the present time in support of policy development.

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

The total figure can be divided by the total area of damaged forest according to the forest damage inventory to provide an estimate of cost per hectare over a five year period. Multiplying this by the incremental increase in forest damage area due to operation of the fuel cycle provides a lower estimate of damages, assuming that such measures would be applied. The assessment provides a lower boundary because the analysis is, at the present time, incomplete.

IV.4. Assessment of Eutrophication Effects on Natural Ecosystems

In addition to acidification, inputs of nitrogen may cause an eutrophication of ecosystems. Too high nitrogen inputs displace other important nutrients or impair their take-up (Matzner and Murach, 1995). This causes nutrient imbalances and deficiency symptoms. When the deposited nitrogen is not completely used for primary production, the excess nitrogen can be inactively accumulated in the system, washed out or emitted again as nitrous oxide (N_2O). Furthermore, the competition between different populations of organisms is influenced, at the expense of species which have evolved to dominate in nutrient poor soils (Nilsson and Grennfelt, 1988; Breemen and Dijk, 1988; Heil and Diemont, 1983). Accordingly, the *critical load for nitrogen nutrient effects* is defined as

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

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

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

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

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

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

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

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

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

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

where

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

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

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

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V. ASSESSMENT OF GLOBAL WARMING DAMAGES

V.1. Introduction

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

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

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

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

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

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

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

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

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

V.2. Interpretation of Results

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

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

In view of these problems, and in the interests of providing policy makers with good guidance, the task team has sought (though inevitably within limits) to avoid introducing personal bias on issues like discount rate, which could force policy in a particular direction. There is a need for other users of the results, such as energy systems modellers or policy makers, to both understand and pass on information regarding uncertainty, and not to ignore it because of the problems that inevitably arise. The Project team 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.

V.3. Discounting Damages Over Protracted Timescales

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

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

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

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

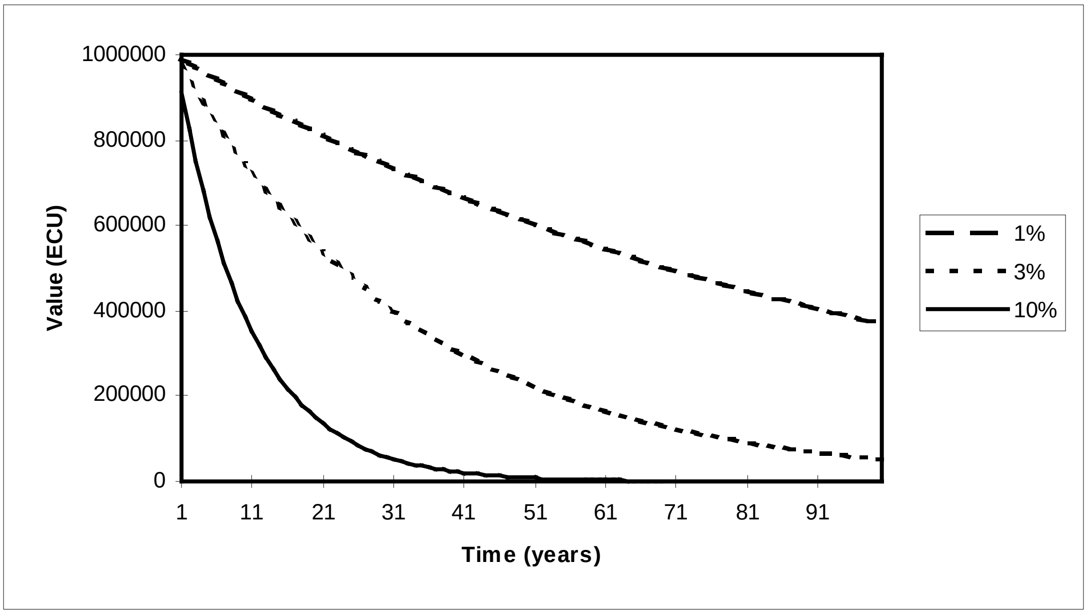

Figure V-1 *Effect of discount rate on present value of damages worth 1 million ECU at the time (from 1 to 100 years in the future) when damage is incurred.*

Given the nature of the ExternE project, some consideration of other types of damage is important as a check on consistency. The most extreme example concerns the consequences of long term disposal of high level radioactive waste. These are commonly assessed over periods of 10,000 years or more. The use of any discount rate more than marginally above zero would reduce damages to a point at which they would be considered negligible in a fraction of this time. Even using a rate of 1%, any damage occurring in 10,000 years time would need to be divided by a factor of $1.6x10^{43}$ to obtain present value. The simple fact that such extended time-spans are considered necessary for assessment of some forms of environmental damage suggests that policy makers do not consider traditional economic analysis to apply in the long term.

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

V.4. Results

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

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

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

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

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

- 1990 ECU:1990 US$ currency conversion - a factor of 0.8,
- 1995 ECU: 1990 ECU consumer price index inflation - a factor of 1.2, and
- revaluation for a 1995 start year - a factor of 1.05 at a 1% discount rate, 1.15 at 3%.

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

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

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

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

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

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

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

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

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

V.5. Conclusions

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

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

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

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

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

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VI. VALUATION ISSUES

VI.1. Introduction

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

The following issues are covered;

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

VI.2. Techniques

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

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

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

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

VI.3. Categories of Value

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

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

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

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

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

VI.4. Transferability of Valuation Data

VI.4.1. Benefit Transfer

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

a) original data sets vary from those in the place of application, and the problems inherent in non-market valuation methods are magnified if transferring to another area;

b) monetary estimates are often stated in units other than the impacts. For example, in the case of damage by acidic deposition to freshwater fisheries, dose response functions may estimate mortality (reduced fish populations) while benefit estimates are based on behavioural changes (reduced angling days). The linkage between these two units must be established to enable damage estimation;

c) studies most often estimate benefits in average, non-marginal terms and do not use methods designed to be transferable in terms of site, region and population characteristics.

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

VI.4.2. Expert Opinion

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.

VI.4.3. Meta Analysis

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

VI.4.4. Conclusions on benefit transfer

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

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

VI.5. Estimation of Uncertain and Risky Phenomena

A separate but equally important aspect of the uncertainty dimension in the valuation of environmental impacts arises from the fact that, for the health related damages, one is valuing changes in risk of damage. Thus the health impacts are usually in the 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.

VI.6. Discounting

VI.6.1. Introduction

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

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

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

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

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

VI.6.2. The Discounting Debate from an Environmental Perspective

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

a) pure time preference;

b) social rate of time preference;

c) opportunity cost of capital;

d) risk and uncertainty;

e) the interests of future generations.

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

VI.6.2.1 Pure Individual Time Preference

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

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

VI.6.2.2 Social Rate of Time Preference

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

$$i = ng + z$$

where z is the rate of pure time preference, g is the rate of growth of real consumption per capita, and n is the percentage fall in the *additional* utility derived from each percentage increase in consumption (n is referred to as the 'elasticity of the marginal utility of consumption'). A typical value for n would be one. With no growth in per capita consumption, the social rate of time preference would be equal to the private rate, z. If consumption is expected to grow the social rate rises above the private rate. The intuitive rationale here is that the more one expects to have in the future, the less one is willing to sacrifice today to obtain even more in the future. Moreover, this impact is greater the faster marginal utility falls with consumption.

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

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

VI.6.2.3 Opportunity Cost of Capital

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

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

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

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

VI.6.2.4 Risk and Uncertainty

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

- uncertainty about whether an individual will be alive at some future date (the 'risk of death' argument),

- uncertainty about the preferences of the individual in the future, and

- uncertainty about the size of the benefit or cost.

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

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

The third kind of uncertainty is relevant, but the difficulty is in allowing for it by adjusting the discount rate. Such adjustments assume that the scale of risks is increasing exponentially over time. Since there is no reason to believe that the risk factor takes this particular form, it is inappropriate to correct for such risks by raising the discount rate. 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.

VI.6.2.5 The Interests of Future Generations

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

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

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

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

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

VI.6.3. Discount Rates and Irreversible Damage

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

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

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

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

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

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

VI.6.4. A Sustainability Approach

The environmental debate has undoubtedly contributed to valuable intellectual soul-searching on the rationale for discounting. But it has not been successful in demonstrating a case for rejecting discounting as such. This Section began by examining the concern over the use of discount rates which reflect pure time preference, but concluded that this concern does not provide a case for rejecting pure time preference completely. However, it was noted that an abnormally high time preference rate 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.

VI.6.5. Conclusions

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

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

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

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VII. UNCERTAINTY AND SENSITIVITY ANALYSIS

VII.1. Introduction

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

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

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

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

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

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

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

VII.2. Analysis of Statistical Uncertainty

VII.2.1. Basis for the Analysis of Uncertainty

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

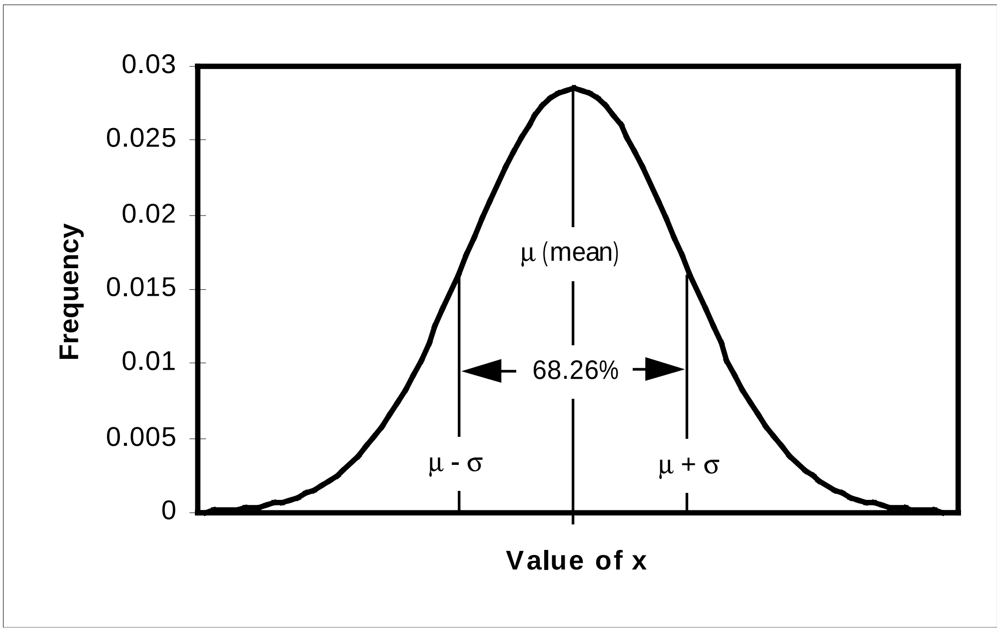

Figure VII-1 *Illustration of the normal distribution.*

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

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

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

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

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

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

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

- Estimation of emissions - for the macropollutants this is the best quantified stage of the analysis, with errors typically of the order of a few percent only.

- Dispersion - established models are available for describing the dispersion of pollutants around a point source, or from a number of different sources. The models are complex needing to integrate chemical processes and variations in meteorology over the extended distances over which they need to be applied. Overall, these models seem reasonably reliable, though it is difficult to validate output, and they are typically incapable of dealing with fine scale variation in pollution climate.

- Dose-response function - a number of epidemiological studies are available for assessing the acute effects of exposure to fine particles on mortality. Results are generally consistent.

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

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

the geometric standard deviation of the physical damage is σ_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 μ_g = 2 deaths/year, the one σ_g interval is 2/2.7 = 0.74 to 2*2.7 = 5.4 deaths/year, and the 95% confidence interval is $2/2.7^2$ = 0.27 to $2*2.7^2$ = 14.58 deaths/year. This result provides an indication of the likely range of outcomes based on statistical uncertainties, and an illustration of the shape of the probability distribution, skewed to the left, but with a long tail going out to high values.

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

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

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

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

VII.2.2. Confidence Bands

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

A = high confidence, corresponding to σ_g = 2.5 to 4;

B = medium confidence, corresponding to σ_g = 4 to 6;

C = low confidence, corresponding to σ_g = 6 to 12;

These bands are reported impact by impact elsewhere within this report. Given that σ_g has actually been quantified for a number of impacts (as in Table VII-1), it is reasonable to ask why the final result is given as a band. The reason is that the data given in this section are themselves uncertain. To give a single figure would imply greater confidence in the characterisation of uncertainty than really exists.

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

VII.3. Key Sensitivities

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

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

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

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

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

VII.4. Conclusions

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

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

- it provides a means of integrating information across disciplines

- results emerge at all stages of the impact pathway providing estimates for example of emission, population exposure, and extent of impacts, as well as monetary damages.

- the use of money for quantification of the final results provides an easily understood weighting system based on public preference.

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

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

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VIII. COAL FUEL CYCLE, FINLAND

VIII.1. Appendix tables of the coal fuel cycle

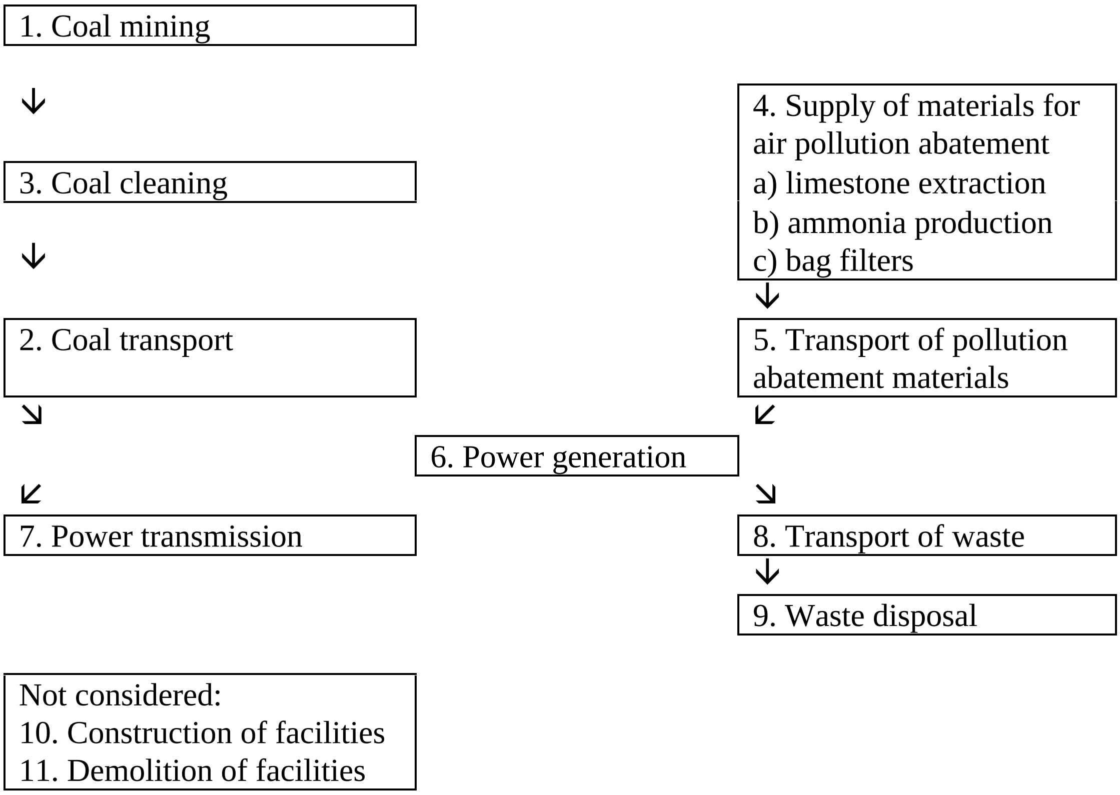

Figure VIII-1 *Coal fuel cycle.*

***Table VIII-1**. Detailed definition of the coal fuel cycle analysed in this study.*

Stage	Parameter	Quantity	Source of data, comments
1. Coal mining	Location(s)	Upper Silesia, Poland	
	Type of mine	Underground	
	Calorific value of coal	25.2 MJ/kg	Design value of plant
	Mine air quality control	not specified	
	Control of mine methane emissions	none	
	Mine waste disposal site	not specified	
	Composition of coal		
	water	8 - 9 %	Ekono Energy (1996)
	ash	12 - 14 %	
	carbon	74 %	
	oxygen	not specified	
	hydrogen	not specified	
	sulphur	0.6 - 0.7 %	
	nitrogen	not specified	
	chlorine	not specified	
	trace elements	not specified	
2. Coal transport	Distance to power station	1450 km	
	Mode of transport	Rail - 550 km Ship - 900 km	
	Rail load	2600 t	Doyle (1989)
	Ship load	45000 t	Default value for av. vessel size
	Nr of rail loads per yr	460	
	Nr of ship loads per yr	27	
3. Coal cleaning	Processes adopted	washing plant: dense medium, magn.separation; jigs	
4. Extraction, production of pollution abatement materials			
4a. Limestone	Mine location	Gotland, Estonia	
	Production	Finland	
5. Transport of pollution abatement materials			
5a. Limestone	Distance to power station	200 km	
	Mode of transport	Lorry	
	Lorry load	35 t	
	Number of lorries per yr	997	
6. Power generation	Fuel	coal	
	Type of plant	pulverised fuel	
	Name	Meri-Pori	
	Location	Pori, Tahkoluoto	
	Geographical latitude	61.63°	
6. Power generation (continued)	Geographical longitude	21.41°	
	Power generation		
	gross	590 MW	
	sent out	560 MW	
	Efficiency	43.1 % (HHV basis)	

	Load factor	74 %	
	Lifetime	?	
	Pollution control		
	ESPs	99.5% effective	
	low NO_x burners	70 % reduction	
	phased combust.		
	SCR(sel.cat.red.)		
	desulphurisation	90 % reduction	
	Stack parameters	(double-stack)	
	height	156 m	Imatran Voima (1995)
	diameter	2×3.7 m	Koski (1996)
	flue gas volume	1656000 Nm^3/h	
	flue gas temp.	393 K	
	Material demands		
	coal	1,195,000 t/a	
	limestone	34,800 t/a	
	cooling water	14.5 m^3/s	Imatran Voima (1995)
7. Transmission	Length of new lines	0 km	
8. Transport of waste	Mode of transport	lorry	
9. Waste disposal	Type of facility	landfill	
10. Construction of facilities		not quantified	
11. Demolition of facilities		not quantified	

Table VIII-2 *Quantification of burdens.*

Stage	Burden		Quantity	Source of data	Impact assessed?
1. Coal mining (outside EU)	Occupational health				
		accidents - fatal	0.59/Mt coal	ILO (1995)	✓
		injuries + occup. diseases	22.95/Mt coal	ILO (1995)	✓
	Air emissions			Ekono Energy (1996)	
		CO_2	59,900 t/a		✓
		CH_4	10,200 t/a		✓
		SO_2	532 t/a		✓
		NO_x	104 t/a		✓
		TSP	56 t/a		✓
	Other burdens				
		mine drainage	not quantified		0 - internalised
		solid wastes	not quantified		x - no data
		subsidence	not quantified		0 - internalised
		noise	not quantified		x - negligible
2. Coal transport (outside EU)	Occupational health and public health (including e.g. train accidents in Poland and ship accidents)		not quantified		
	Air emissions (includes also transport of stages 5 & 8)			Ekono Energy (1996)	
		CO_2	12,500 t/a		✓
		SO_2	50 t/a		✓
		NO_x	270 t/a		✓
		TSP - combustion	19 t/a		✓
		fugitive dust (portion of TSP unknown)	1400 t/a		not assessed
	Other burdens				
		noise	not quantified		
		burden on infrastructure	not quantified		
3. Coal cleaning	burdens included in other stages				
4. Extraction, production of pollution abatement materials					
4a. Limestone	Occupational health		not quantified		
	Air emissions		not quantified		
	Emissions to water		not quantified		
	Other burdens		not quantified		
5. Transport of pollution abatement materials (inside EU)					
5a. Limestone	Occupational health		not quantified		
	Public health (lorry accidents, includes also stage 8)				
		accidents - fatal	0.029 accid./a		✓
		accidents - major	0.12 accid./a		✓

Stage	Burden	Quantity	Source of data	Impact assessed?
	injury			
	Air emissions (included in stage 2)			✓
	Noise	not quantified		
	Burden on infrastructure	not quantified		
6. Power generation	Occupational health	not quantified		
	Air emissions			
	CO_2	2,802,000 t/a		✓
	N_2O	60 t/a		✓
	CH_4	150 t/a		✓
	SO_2	4000 t/a		✓
	NO_x	1900 t/a		✓
	TSP	540 t/a*		✓
	fugitive dust from coal storage of power plant	4500 t/a		impact not assessed
	trace elements	not quantified		
	Noise emissions	45 dB(A) at 100 m distance		
	Solid waste production			
	ash	150,000 t/a	Imatran Voima (1995)	
	gypsum	60,000 t/a	Imatran Voima (1995)	
	Water abstraction			
	Emissions to water from. cooling system			
	temperature increase of cooling water on return	10 °C		
*this number appears to be an overestimate				
7. Transmission	No additional burdens			
8. Transport of waste (inside EU)	Occupational health	not quantified		
	Public health (traffic accidents included in stage 5)			✓
	Air emissions (included in stage 2)			✓
	Noise	not quantified		
	Burden on infrastructure	not quantified		
9. Waste disposal	Occupational health	not quantified		
	Air emissions	not quantified		
	Leachate	not quantified		
	Quantity of waste	not quantified		
	Noise	not quantified		
	Road use	not quantified		

Stage	Burden	Quantity	Source of data	Impact assessed?
10. Construction	Occupational health	not quantified		
	Air emissions from materials production	not quantified		
	Air emissions from materials transport	not quantified		
	Air emissions from activities on site	not quantified		
	Noise	not quantified		
	Road use	not quantified		
11. Demolition	Occupational health	not quantified		
	Air emissions from materials production	not quantified		
	Noise	not quantified		
	Road use	not quantified		

Table VIII-3 Impacts and damages of the coal fuel cycle.

Impact	Impact - units/TWh	Impacts - number	Damages mECU/kWh	ECU/t_{poll} (VOLY applied)	σ_g, range
Air pollution					
6. Power generation					
Primary TSP				**1555**	
Acute mortality	VOLY deaths	not considered			
Chronic mortality	VOLY deaths	2.4 0.24	0.2 0.7	1378	
Acute and chronic morbidity	cases	151	0.03	178	
Materials - cleaning costs		not considered			
NO_x - all from nitrate aerosol				**1310**	
Acute mortality	VOLY deaths	not considered			
Chronic mortality	VOLY deaths	7.3 0.73	0.4 2.3	1160	
Acute and chronic morbidity	cases	457	0.08	150	
Materials damage - utilitarian		not considered			
Materials damage - cultural		not considered			
Critical loads exceedance	km^2	162	not quantified		
Effects on crop yield	kg fertiliser added	-3308	-6E-5	-0.1	
SO_2 - as sulphate, unless marked SO_2				**1486**	
Acute mortality (SO_2)	VOLY deaths	0.16 0.02	0.025 0.05	38	
Chronic mortality	VOLY deaths	9.8 0.98	0.83 3.1	1240	
Acute and chronic morbidity	cases	623	0.1	155	
Materials damage - utilitarian	m^2 maint. area	2435	0.03	48	
Materials damage - cultural		not considered			
Critical loads exceedance SO_2	km^2	42	not quantified		
Effects on crop yield	dt yield loss	444	0.003	5	
O_3				**1500**	
Acute mortality	VOLY deaths		0.2	415	
Acute morbidity	cases		0.4	735	
Materials damage - utilitarian		not considered			
Critical loads exceedance	km^2	not considered			
Effects on crop yield			0.2	350	
Greenhouse gas emissions					
CO_2	t	770,000	2.3	3	low
CO_2			12	16	mid 3%
CO_2			35	46	mid 1%
CO_2			107	139	high
N_2O	t	17	0.02	930	low
N_2O			0.1	4960	mid 3%
N_2O			0.2	14260	mid

N_2O			0.7	43090	1% high
Other stages - within Europe (stages 1, 2, 5 and 8)					
Primary TSP				**1555**	
Acute mortality	VOLY deaths	not considered			
Chronic mortality	VOLY	0.34	0.03	1378	
	deaths	0.034	0.1		
Acute and chronic morbidity	cases	21	0.004	178	
Materials damage - cleaning costs		not considered			
NO_x - all from nitrate aerosol				**1310**	
Acute mortality	VOLY deaths	not considered			
Chronic mortality	VOLY	1.4	0.12	1160	
	deaths	0.14	0.4		
Acute and chronic morbidity	cases	88	0.015	150	
Materials damage - utilitarian		not considered			
Materials damage - cultural		not considered			
Critical loads exceedance	km^2	31	not quantified		
Effects on crop yield	kg fertiliser added	-636	-1E-5	-0.1	
SO_2 - as sulphate, unless marked SO_2				**1486**	
Acute mortality	VOLY	0.04	0.006	38	
	deaths	0.004	0.012		
Chronic mortality	VOLY	2.4	0.2	1240	
	deaths	0.24	0.7		
Acute and chronic morbidity	cases	149	0.025	155	
Materials damage - utilitarian	m^2 maint. area	583	0.008	48	
Materials damage - cultural		not considered			
Critical loads exceedance	km^2	10	not quantified		
Effects on crop yield	dt yield loss	106	0.0007	5	
O_3				**1500**	
Acute mortality	VOLY deaths		0.04	415	
Acute morbidity	cases		0.08	735	
Materials damage - utilitarian					
Critical levels exceedance	km^2	not considered			
Effects on crop yield			0.04	350	
Greenhouse gas emissions					
CO_2	t	25,000	0.08	3	low
			0.4	16	mid 3%
			1.2	46	mid 1%
			3.5	139	high
N_2O	t	not considered			
CH_4	t	2,800	0.18	63	low
			0.95	336	mid 3%
			2.7	966	mid

			8.2	2920	1% high
Occupational health effects					
1. Coal mining - Poland					
Fatalities	deaths	0.2	0.6		
Major injuries	cases	6.3	0.6		
Minor injuries	cases				
Public health effects (road accidents)					
Stages 5 and 8 - Finland					
Fatalities	deaths	0.008	0.025		
Major injuries	cases	0.03	0.003		
Minor injuries	cases				
Noise					
All stages	-	negligible			

VIII.2. References

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IX. PEAT FUEL CYCLE, FINLAND

IX.1. Appendix tables of the peat fuel cycle

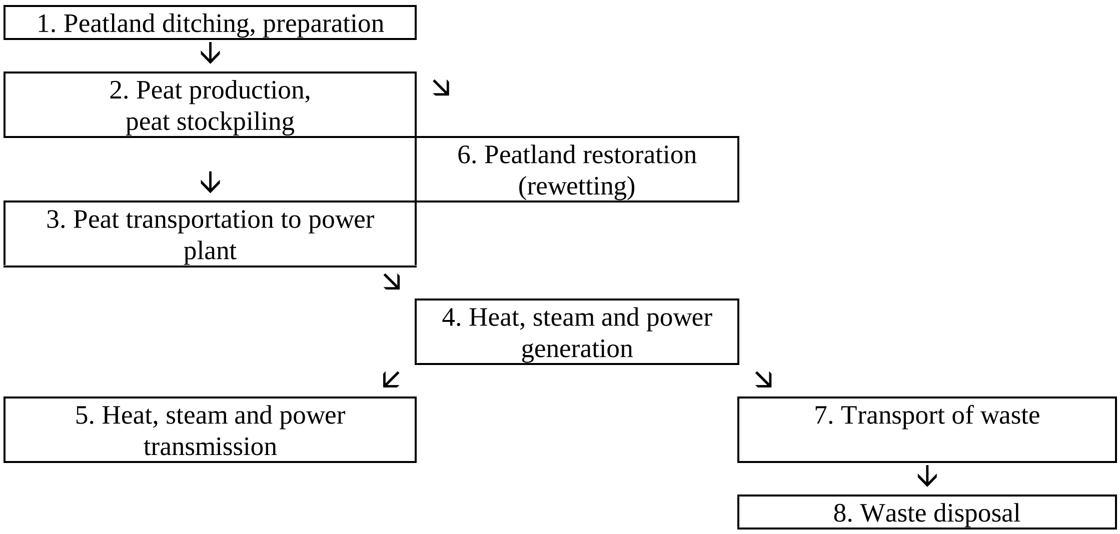

Figure IX-1 *Stages of the peat fuel cycle.*

Table IX-1 *Definition of the peat fuel cycle.*

Stage	Parameter	Quantity	Source of data, comments
1. Peatland ditching, preparation	Location(s) Type of peatland Time of preparation & ditching	Jyväskylä area, FI Natural peatland 3 - 6 years	Mälkki and Frilander, (1997).
2. Peat production	Type of peat Production method	Milled peat Tehoturve	Mälkki and Frilander, (1997).
3. Peat transportation	Distance to power station Mode of transport	80 km Road - 100%	Mälkki and Frilander, (1997).
4. Power generation	Fuels Type of plant Location Calorific value of peat	milled peat, wood residues, coal, oil fluidised bed combustion Jyväskylä, FI 10.1 MJ/kg	Rauhalahti Power Plant

Stage	Parameter	Quantity	Source of data, comments
	Power generation		
	gross	87 MW	
	sent out	83 MW	
	Lifetime	40 years	
	Size of plant		
	land area required		
	height of stack	130 m	
	diameter of stack	3.5 m	
	Other characteristics		
	flue gas temperature	395 K	
	flue gas volume	410,400 Nm^3/h	
	Full load hours	5655 hours/a	calculated for ExternE
	Operating hours 1995	8328 hours/a	Rauhalahti Power Plant
5. Heat, steam and power transmission	District heat transmission	to the city	
		to the local paper mill	
	Steam transmission		
	Power transmission	commercial power distribution network	
6. Peatland restoration	Time of assessed restoration	100 years	Mälkki and Frilander (1997).
	Mode of restoration	rewetting	
7. Transport of waste	Site	Local	Rauhalahti Power Plant
8. Waste disposal	Type of facility	Landfill	Rauhalahti Power Plant
Transport of pollution abatement materials	Service of electrostatic precipitator	not applied	
Construction of facilities	Not applied		
Demolition of facilities	Not applied		

Table IX-2 *Quantification of burdens of the peat fuel cycle.*

Stage	Parameter		Quantity	Source of data, comments
1. Peatland ditching, preparation				Mälkki and Frilander (1997).
	Air emissions			
		CO_2	**6000 t/a**	
		CH_4	**-30 t/a**	
	Water emissions			
		COD	**40 t/a**	
		solids	**30 t/a**	
		tot. N	**3 t/a**	
2. Peat production				Mälkki and Frilander (1997).
	Air emissions			
		CO_2	**54000 t/a**	
		CH_4	**-160 t/a**	
	Water emissions			
		COD	**140 t/a**	
		solids	**70 t/a**	
		tot. N	**10 t/a**	
3. Peat transportation				Mälkki and Frilander (1997).
	Air emissions			
		SO_2	**3 t/a**	
		NO_x	**40 t/a**	
		CO_2	**3000 t/a**	
4. Power generation				Rauhalahti Power Plant
	Calorific value of peat		**10.1 MJ/kg**	
	Fuels used			
		milled peat	**520000 t/a**	
		coal	**1900 t/a**	
		wood chips	**94000 t/a**	
		oil	**430 t/a**	
	Emissions	SO_2	**1300 t/a**	
		NO_x	**760 t/a**	
		CO_2	**630000 t/a**	
		TSP	**90 t/a**	
5. Heat, steam and power transmission				

6. Peatland restoration	Assessed restoration time		**100 years**	
	Mode of restoration		**rewetting**	Mälkki and Frilander (1997).
	Air emissions	CO_2	**-35000 t/a**	
		CH_4	**-400 t/a**	
7. Transport of waste	Site		**Local**	Rauhalahti Power Plant
8. Waste disposal	Type of facility		**Landfill**	Rauhalahti Power Plant
Transport of pollution abatement materials	Service of electrostatic precipitator		**Not applied**	
Construction of facilities	Not applied			
Demolition of facilities	Not applied			
Total life cycle of peat utilisation	Including stages 1-8 and fuel chains of diesel, oil, coal and wood chips			Mälkki and Frilander (1997).
	Total emissions			
		SO_2	**1300 t/a**	
		NO_x	**800 t/a**	
		TSP	**100 t/a**	
		CO_2	**660000 t/a**	
		CH_4	**-600 t/a**	
		N_2O	**30 t/a**	

Table IX-3 *Impacts and damages of the peat fuel cycle.*

Impact	Impact - units/TWh	Impacts - number	Damages mECU/kWh	ECU/tpoll	sg, range
Air pollution					
6. Power generation					
Primary TSP				1343.6	
Acute mortality	VOLY	not considered	?		
	deaths				
Chronic mortality	VOLY	1.11	0.093		
	deaths	0.111			
Acute and chronic morbidity	cases	69	0.0121		
Materials - cleaning costs		not considered			
NOx - all from nitrate aerosol				856	
Acute mortality	VOLY	not considered			
	deaths				
Chronic mortality	VOLY	9.4	0.790		
	deaths	0.94			
Acute and chronic morbidity	cases	584	0.102		
Materials damage - utilitarian		not considered			
Materials damage - cultural		not considered			
Critical loads exceedance	km2	286			
Effects on crop yield		negligible			
SO_2 - as sulphate, unless marked SO_2				1027	
Acute mortality	VOLY	0.09	0.280		
	deaths	0.02?			
Chronic mortality	VOLY	12	1.07		
	deaths	1.247058			
Acute and chronic morbidity	cases	762	0.13		
Materials damage - utilitarian			0.028		
Materials damage - cultural		not considered			
Critical loads exceedance	km^2	72			
Effects on crop yield			0.0045		
O_3					
Acute mortality	VOLY		0.45	415	
	deaths				
Acute morbidity	cases		0.80	735	
Materials damage - utilitarian		not considered			
Critical loads exceedance	km^2	not considered			
Effects on crop yield			0.38	350	
Greenhouse gas emissions					
CO2	t	971904	2.9	3	low
			15.6	16	mid 3%
			44.7	46	mid 1%
			135	139	high
N2O	t	0	0.000	930	low

				4960	mid 3%
				14260	mid 1%
				43090	high
Other stages - within Europe (stages 1, 2, 5 and 8)					
Primary TSP				1344	
Acute mortality	VOLY	not considered			
	deaths				
Chronic mortality	VOLY	0.129	0.0109		
	deaths	0.0129			
Acute and chronic morbidity	cases	8.1	0.0014		
Materials damage - cleaning costs		not considered			
NOx - all from nitrate aerosol				856	
Acute mortality	VOLY	not considered			
	deaths				
Chronic mortality	VOLY	0.78	0.066		
	deaths	0.08			
Acute and chronic morbidity	cases	48	0.008		
Materials damage - utilitarian		not considered			
Materials damage - cultural		not considered			
Critical loads exceedance	km^2	24			
Effects on crop yield		negligible			
SO2 - as sulphate, unless marked SO2				1027	
Acute mortality	VOLY	0.001	0.0043		
	deaths	?			
Chronic mortality	VOLY	0.19	0.02		
	deaths	0.019			
Acute and chronic morbidity	cases	12	0.002		
Materials damage - utilitarian			0.000		
Materials damage - cultural		not considered			
Critical loads exceedance	km2	1.1			
Effects on crop yield			0.000069		
O3					
Acute mortality	VOLY		0.038	415	
	deaths				
Acute morbidity	cases		0.067	735	
Materials damage - utilitarian		not considered			
Critical levels exceedance	km^2	not considered			
Effects on crop yield			0.032	350	
Greenhouse gas emissions					
CO_2	t	41182	0.12	3	low
			0.66	16	mid 3%
			1.9	46	mid 1%
			5.7	139	high
N_2O	t	35.01	0.03	930	low
			0.17	4960	mid 3%
			0.50	14260	mid 1%
			1.5	43090	high
CH_4	t	-803.06	-0.05	63	low
			-0.27	336	mid 3%

			-0.78	966	mid 1%
			-2.3	2919	high

Public health effects (road accidents) Stage 3 - Finland

Fatalities	deaths	0.186	0.394
Major injuries	cases	0.743	0.048

Noise

All stages	-	not considered

IX.2. References

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X. BIOMASS FUEL CYCLE, FINLAND

X.1. Appendix tables of the biomass fuel cycle

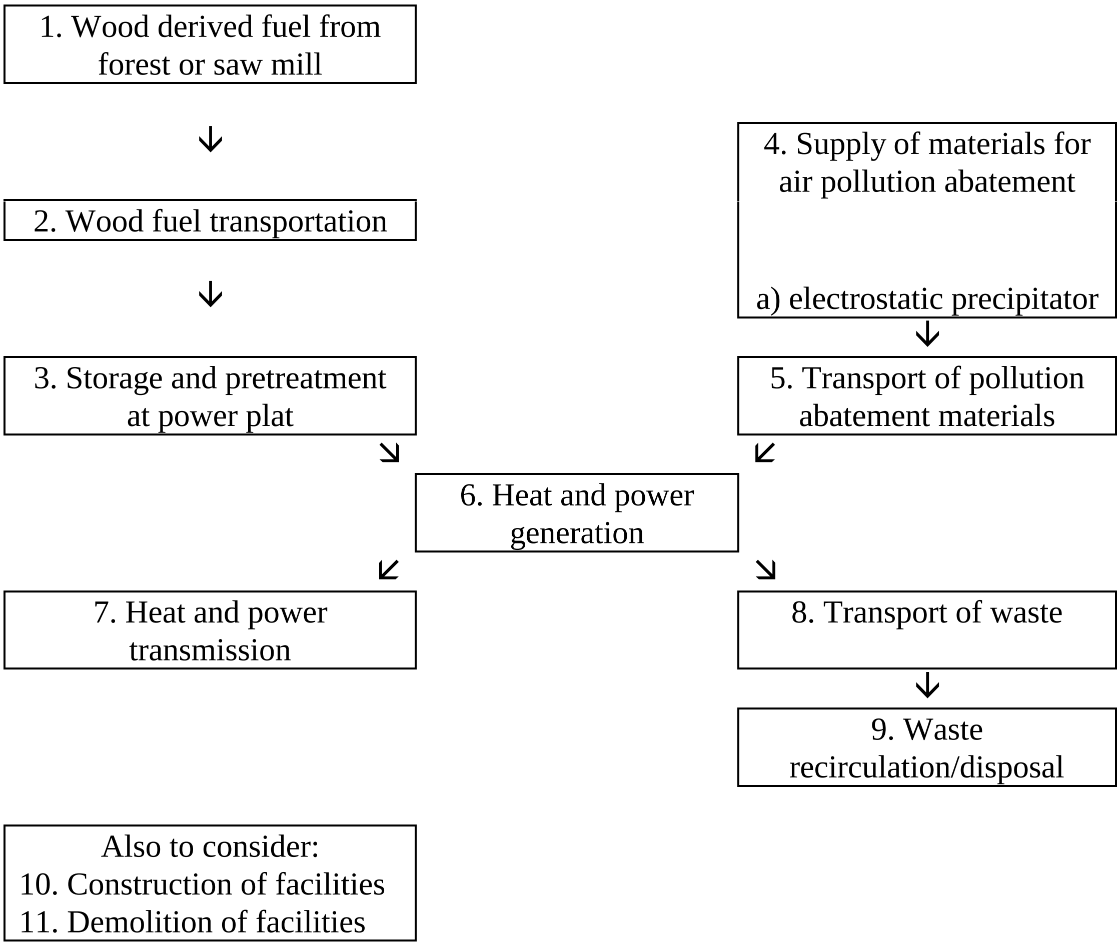

Figure X-1 *Biomass fuel cycle.*

Table X-1 *Detailed definition of the biomass fuel cycle analysed in this study.*

Stage	Parameter	Quantity	Source of data, comments
1. Wood fuel production	**Fuel need**	Total 260 GWh/a	Forssan Energia Oy 1996
	Fuel chips from forestry	50 000 m^3 0.8 MWh/m^3	estimation/Biowatti Oy 1996
	Bark from saw mill	185 000 m^3 0.7 MWh/m^3	estimation/Biowatti Oy 1996
	Saw dust from saw mill	130 000 m^3 0.55 MWh/m^3	estimation/Biowatti Oy 1996
	Other wood waste	<35 000 m^3 0.8 MWh/m^3	estimation/Biowatti Oy 1996
	Fuel properties		
	moisture	40 - 55 %	
	ash	0.4 - 0.8 %	of dry matter
	calorific value	19-20 MJ/kg	of dry matter
	sulphur content	0.05 %	of dry matter
	Energy use for fuel chips production		
	- wood chips	46 t diesel oil	Chipset/VTT 1996
	- saw dust	0	by product (waste)
	- bark	330 MWh_{el}	chipping
	- other wood waste	0	by product (waste)
	Emissions		
	- Diesel fuel		
	SO_2	4 g/l	0.22 t/a
	NO_x	50 g/l	2.70 t/a
	CO_2	2.7 kg/l	146 t/a
	TSP	5 g/l	0.27 t/a
	CO	15 g/l	0.81 t/a
	N_2O	0.1 g/l	0.006 t/a
	CH_4	0.4 g/l	0.02 t/a
	- Electricity		
	SO_2	650 g/MWh_{el}	0.22 t/a
	NO_x	650 g/MWh_{el}	0.22 t/a
	CO_2	270 kg/MWh_{el}	90 t/a
	TSP	150 g/MWh_{el}	0.05 t/a
	CO	800 g/MWh_{el}	0.27 t/a
	N_2O	5 g/MWh_{el}	0.002 t/a
	CH_4	90 g/MWh_{el}	0.03 t/a
2. Wood fuel transportation	Truck	0 - 100 km	
	Energy use from truck	51 l/100 km	(300 000 km)
	Truck emission		
	SO_2	4 g/l	0.61 t/a
	NO_x	52 g/l	7.96 t/a
	CO_2	2.7 kg/l	413 t/a
	TSP	3 g/l	0.46 t/a
	CO	9 g/l	1.38 t/a
	N_2O	0.1 g/l	0.015 t/a
	CH_4	0.4 g/l	0.06 t/a

Stage	Parameter		Quantity	Source of data, comments
3. Storage and pretreatment at power plant	Fuel storage		800 + 150 m^3	Impola , R. 1997
	Stockpiles			no average data
4. Supply of materials for air pollution abatement	Electrostatic precipitator		not applied	negligible
5. Transport of pollution abatement materials	Service of electrostatic precipitator		not applied	negligible
6. Heat and power generation	Fuel			
	Wood derived biomass		260 GWh/a	O. Kaulamo Engineering 1996
	Light fuel oil		10 m^3/a (0.1 GWh)	IVO International 1994
	Type of plant		Co-generation bubbling fluidised bed	O. Kaulamo Engineering 1996
	Fuel effect		72 MW	O. Kaulamo Engineering 1996
	Electrical output		17.2 MW, 56.8 GWh/a	O. Kaulamo Engineering 1996
	Heat output		48 MW, 155 GWh/a	O. Kaulamo Engineering 1996
	Full load hours		3600 h/a	O. Kaulamo Engineering 1996
	Fuel gas volume		30 m^3(n)/s	O. Kaulamo Engineering 1996
	Fuel gas temperature		403 K	O. Kaulamo Engineering 1996
	Stack height		50 m	O. Kaulamo Engineering 1996
	Over sea level		178.5 m	Forssan Energia Oy 1996
	Stack diameter		1.6 m	O. Kaulamo Engineering 1996
	Anemometer height		60 m	
	Latitude		60.79°	Forssan energia Oy 1996
	Longitude		23.59°	Forssan Energia Oy 1996
	Elevation		128 m	
	Emissions	SO_2	40 mg/MJ, biomass*	O. K. E 1996
		NO_x	150 mg/MJ*	O. K. E 1996, 140 t/a , part of this is probably ”neutral”
		CO_2	(121 g/MJ, biomass)	O. K. E 1996, (113 000 t/a), part of natural carbon cycle
			74 g/MJ (POK)	O. K. E 1996
		TSP	20 mg/MJ	O. K. E 1996
		CO	100 mg/MJ	O. K. E 1996
		N_2O	20 mg/MJ	O. K. E 1996
		CH_4	40 mg/MJ	Boström et al 1992
	Lifetime		25 - 30 years	Impola, R. 1997

***The actual emissions appear to be essentially smaller: SO_2 nearly 0 and NO_x 80-90 mg/MJ**

7. Heat and power transmission	District heat transmission distance Power transmission	3.5 km comercial power-distribution network	IVO International 1994
8. Transportation of waste	Ashes (pure)	500 - 1000 t	
	Bottom ashes	10 %	IVO International 1994
	Fly ashes	90 %	
	Transport distance	< 1 km	IVO International 1994
9. Waste disposal	Recycling/landfill	Landfill 100 %	IVO International 1994, recycling probability
10. Construction of facilities	Not applied		
11. Demolition of facilities	Not applied		

O.K.E =Osmo Kaulamo Engineering

Table X-2 *Quantification of burdens.*

Stage	Burden	Quantity	Source of data	Impact assessed?
1. Wood fuel prod.	Occupational health			
	accidents - fatal	0.000 000 05 /$MWh_{wood\ fuel}$	Forest work, Metsätilastollinen vuosikirja 1996	✓
	accidents - injury	0.000 02 /$MWh_{wood\ fuel}$	Forest work, Metsätilastollinen vuosikirja 1996	✓
	Air emissions			
	- Diesel fuel			
	SO_2	0.22 t/a		✓
	NO_x	2.70 t/a		✓
	CO_2	1500 t/a		✓
	TSP	0.27 t/a		✓
	CO	0.81 t/a		
	N_2O	0.006 t/a		✓
	CH_4	0.02 t/a		✓
	- Electricity			
	SO_2	0.22 t/a		✓
	NO_x	0.22 t/a		✓
	CO_2	90 t/a		✓
	TSP	0.05 t/a		✓
	CO	0.27 t/a		
	N_2O	0.002 t/a		✓
	CH_4	0.03 t/a		✓
	Other burdens			
	noise	not quantified		x-negligible
	ecological	not quantified		x - negligible
	coal balance in soil	not quantified		x - negligible
2. Biomass transport.	Occupational and public health			
	accidents - fatal	0.000 000 01 /$MWh_{wood\ fuel}$	Traffic statistics, Suomen tilastollinen vuosikirja 1996	✓
	accidents - injury	0.000 000 2 /$MWh_{wood\ fuel}$	Traffic statistics, Suomen tilastollinen vuosikirja 1996	✓
	Air emissions			
	Truck emission			
	SO_2	0.61 t/a		✓
	NO_x	2.92 t/a		✓
	CO_2	240 t/a		✓
	TSP	0.32 t/a		
	CO	1.08 t/a		✓

Stage	Burden	Quantity	Source of data	Impact assessed?
	N_2O	0.001 t/a		✓
	CH_4	0.06 t/a		✓
	Other burdens			
	noise	not quantified		
	burden on infrastructure	not quantified		
3. Storage and pretreatment at power plant	fugitive dust	not quantified		
4. Supply of materials for air pollution abatement		not quantified		x - negligible
5. Transport of pollution abatement materials		not quantified		x - negligible
6. Power generation	Occupational health	not quantified		
	Air emissions			
	SO_2	37 t/a*		✓
	NO_x	140 t/a*		✓
	CO_2	113 t/a		✓
	TSP	19 t/a		✓
	CO	93 t/a		
	N_2O	19 t/a*		✓
	CH_4	37 t/a		✓
	Noise	<45 dB, at plant area border		
	Ash	< 1000 t/a		x - negligible
	Waste water abstraction	50 000 t/a		x - negligible
7. Transmission		not quantified		x - negligible
8. Transport of waste				x - negligible
9. Waste disposal		not quantified		x - negligible
10. Construction		not quantified		
11. Demolition		not quantified		

*these numbers are overestimated.

Table X-3 *Impacts and damages from the biomass fuel cycle.*

Impact	*Impact - units/TWh*	Impacts - number	Damages mECU/kWh	*ECU/t_{poll}*	σ_g, range
Air pollution					
6. Power generation					
Primary TSP				2611	
Acute mortality	VOLY				
	deaths				
Chronic mortality	VOLY	5.69	0.48		
	deaths	0.57			
Acute and chronic morbidity	cases	355	0.06		
Materials - cleaning costs					
NO_x - all from nitrate aerosol				1388	
Acute mortality	VOLY	not considered			
	deaths				
Chronic mortality	VOLY	22.7	1.912		
	deaths	2.27			
Acute and chronic morbidity	cases	1413	0.247		
Materials damage - utilitarian		not considered			
Materials damage - cultural		not considered			
Critical loads exceedance	km^2	433			
Effects on crop yield					
SO_2 - as sulphate, unless marked SO_2				1607	
Acute mortality	VOLY	0.01	0.002		
	deaths	0.001			
Chronic mortality	VOLY	0.56	0.5		
	deaths	0.06			
Acute and chronic morbidity	cases	389	0.0658		
Materials damage - utilitarian			0.048		
Materials damage - cultural		not considered			
Critical loads exceedance	km^2	22			
Effects on crop yield			0.0002		
O_3					
Acute mortality	VOLY		0.65	415	
	deaths				
Acute morbidity	cases		1.14	735	
Materials damage - utilitarian					
Critical loads exceedance	km^2	not considered			
Effects on crop yield			0.54	350	
Greenhouse gas emissions					
CO_2	t	125	0.009	7.4	
N_2O	t	21	0.476	2294	
Other stages - within Europe (stages 1, 2, 5 and 8)					
Primary TSP				2611	
Acute mortality	VOLY				

	deaths			
Chronic mortality	VOLY	0.23	0.02	
	deaths	0.02		
Acute and chronic morbidity	cases	14.63	0.003	
Materials damage - cleaning costs		not considered		
NO_x - all from nitrate aerosol				1388
Acute mortality	VOLY	not considered		
	deaths			
Chronic mortality	VOLY	1.75	0.15	
	deaths	0.17		
Acute and chronic morbidity	cases	109.05	0.02	
Materials damage - utilitarian		not considered		
Materials damage - cultural		not considered		
Critical loads exceedance	km^2	33		
Effects on crop yield		negligible		
SO_2 - as sulphate, unless marked SO_2				1607
Acute mortality	VOLY	0.0004	0.0001	
	deaths	0.00004		
Chronic mortality	VOLY	0.0156	0.0013	
	deaths	0.0016		
Acute and chronic morbidity	cases	10.8	0.0018	
Materials damage - utilitarian			0.0013	
Materials damage - cultural		not considered		
Critical loads exceedance	km^2	0.6		
Effects on crop yield			0.000006	
O_3				
Acute mortality	VOLY		0.050	415
	deaths			
Acute morbidity	cases		0.088	735
Materials damage - utilitarian		not considered		
Critical levels exceedance	km^2	not considered		
Effects on crop yield			0.042	350
Greenhouse gas emissions				
CO_2	t	722	0.05	7.4
N_2O	t	0.03	0.0008	2294
CH_4	t	0.12	0.0002	155
Occupational health effects				
1. Wood fuel production				
Fatalities	deaths	0.002	0.11	
Major injuries	cases	1.00	1.67	
Minor injuries	cases			
Public health effects (road accidents)				
Stages 5 and 8 - Finland				
Fatalities	deaths	0.01	0.41	
Major injuries	cases	0.05	0.05	

Minor injuries	cases	
Noise		
All stages	-	negligible

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