New Elements for the Assessment of External Costs from Energy Technologies

New Ext

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CONTENT

I OBJECTIVE

The supply and use of energy imposes risks and causes damage to a wide range of receptors, including human health, natural ecosystems (flora and fauna) and the built environment. Such damages are to a large extent external costs, as they are not accounted for in the factor costs and thus in the decisions of electricity producers. The existence of external effects in the energy sector (but also other industrial activities) may cause welfare losses and a non-optimal allocation of resources

Within the ExternE projects funded under the JOULE Programme during the 1990s, a detailed bottom-up ‘impact pathway’ (or damage function) approach was developed to quantify external costs from energy conversion resulting from impacts on human health, crop losses, material damage and global warming. The ExternE external costs accounting framework is widely accepted and has been successfully used to support decision making in the field of energy and environmental policy.

However, there are also areas for which a need for further research was identified in previous ExternE phases. Major uncertainties result from uncertainties in the monetary valuation of mortality effects and from the omission of impacts on ecosystems due to global warming and acidification and eutrophication of ecosystems. The formerly existing accounting framework was also criticised for not taking into account the contamination of water and soil. Due to accumulation processes of persistent substances there is a significant potential for long-term effects that were not addressed in previous work. Another source for criticism is the unbalanced treatment of severe accidents, as the current framework is very much focused on accidents in the nuclear fuel chain, while neglecting severe accidents from other energy sources. NewExt as the follow-up of former ExternE phases has therefore focussed on the improvement of the existing framework in four key areas, which are considered as most relevant for the assessment of external costs, and which are expected to be primarily affected by new scientific findings. Thus, the main objective of the project has been to improve the assessment of externalities by providing new methodological elements for integration into the existing external costs accounting framework that reflect the most important new developments in the assessment of external costs.

II INTRODUCTION AND OVERVIEW

To achieve this objective, the update of methodologies focussed on four different areas that are examined each in specific work packages. The project provides

- **an improved methodology for the monetary valuation of mortality impacts from air pollution**

  The monetary valuation of mortality impacts has been identified as the dominant parameter in the assessment of external costs from energy conversion. In the last phase of ExternE it was suggested that the most appropriate methodology for the valuation of mortality impacts is the new approach of 'Value of Life Year Lost' (VOLY) for the valuation of mortality impacts. Since no studies directly focussing on the VOLY have yet been conducted in Europe, such a study has been carried out within the project to provide an empirical basis for this most important single parameter in the accounting framework.

- **valuation of environmental impacts based on preferences revealed in (1) political negotiations (global warming, acidification and eutrophication) and (2) public referenda (global warming).**

  The impact pathway requires estimating the impacts in physical terms and then to value these impacts based on the preferences of the 'common man'. This approach has been successfully applied to e.g. human health impacts, but in other areas this approach cannot be fully applied because data on valuation is missing (acidification and eutrophication of ecosystems) or estimation of all physical impacts is limited (global warming). It is estimated that for those areas a full implementation of the impact pathway approach would require large efforts both in terms of physical science and monetary valuations, efforts that go way beyond ExternE.

  Therefore for these cases, a *second best approach* may be better then having no data, or partial data. In NewExt it has been explored to which extend approaches that elicit *implicit* values in policy decisions can be useful to monetise the impacts of global warming, acidification and eutrophication. Traditional approaches to estimate 'shadow prices' per ton of pollutant cannot be used here because they account for the total impacts and are not additive to ExternE estimates for e.g. public health and because they are not site-specific. Therefore a new approach has been elaborated that uses data on costs and benefits used in the preparation and negotiation of the UN-ECE LRTAP protocol of 1999 and the EU NEC-directive of 2001. This data has been reinterpreted to estimate an *implicit* WTP (willingness to pay) per hectare of ecosystem no longer above critical loads. These values can be further used in combination with estimates of how emissions affect the ecosystems in terms of their exceedance of critical loads.

Second, a similar reasoning has been applied to control of CO_2 emissions. The implicit WTP for meeting the emission limits from the Kyoto protocol is dependent on the policy choices related to the instruments how to achieve these targets.

Third, an innovative approach was developed by deriving an implicit WTP for controlling CO_2 emissions from people’s voting behaviour in referenda related to energy questions in Switzerland.

- **a methodology for the assessment of effects from multi-media (air/water/soil) impact pathways**

  The strong focus of ExternE on airborne pollutants has been criticised, as it neglects the significant environmental impacts from the contamination of water and soil resulting from an energy system's full life cycle. In particular, the human exposure to heavy metals and some important organic substances (e.g. dioxins), which accumulate in water and soil compartments and lead to a significant exposure via the food chain, was not well represented. The project identified priority impact pathways and developed methodologies for the quantification of relevant externalities whose results were compared for validation. The multimedia impacts of toxic metals emitted by power plants turn out not to make a significant contribution to the damage costs.

- **a methodology and a related database for the assessment of externalities from major accidents in non-nuclear fuel chains**

  In previous ExternE work, emphasis was placed on the quantification and valuation of impacts from beyond design basis accidents in the nuclear fuel cycle. However, other fuel chains also show a significant potential for severe accidents (e.g. oil fires or large spills, gas explosions, dam failures). The project reviewed and extended existing database systems on major accidents related to energy conversion activities. Furthermore, for hydro power an approach using elements of Probabilistic Safety Assessment (PSA) was defined and some of its components were elaborated on a limited-scope basis. In a second step, a methodology was developed to estimate external costs from major accidents, thus advancing comparability with the results earlier obtained for beyond design basis accidents in the nuclear fuel chain. This work allows for the first time a consistent and comprehensive assessment of externalities from major accidents in non-nuclear fuel chains.

Of course, these four new methodological elements should be compatible with the existing external costs accounting framework. While it has not been the objective of the project to provide a broad review of current external cost estimates by taking into account the new methodology, some testing of the methodology is required to demonstrate its feasibility. The new methodology has been applied to calculate external costs for a set of reference power plants in Germany, Belgium, France and the United Kingdom, for which technical data have been available from previous ExternE work. The question how these new numbers may affect the major policy conclusions of previous work was addressed. One additional essential factor at this stage was the consideration of some parallel new insights, developments and changes that occurred in the scientific field of external costs in parallel to the NewExt project, e. g. changes of applicable dose-response functions.

This project produced a set of new methodological 'building blocks' for integration into the existing EU external costs accounting framework, rather than a 'stand alone' methodology for the assessment of externalities. The communication and dissemination of the new methodological elements to the current users of the existing accounting framework and the relevant scientific community and the guidance on the use of the new methodological elements have been achieved by carrying out a number of workshops and by setting up a webpage (www.externe.info) within the supporting concerted action DIEM (Dissemination and discussion of the ExternE methodology and results).

According to the structure of the NewExt project, the methodological work on the four work packages has each lead to specific new insights and results. Based on all this work, but also on further updates of baseline data, dose-response functions and the EcoSense software, new calculations have been made for the basic fuel cycles, so that a comparison with the results of the National Implementation phase of ExternE can be done.

The following five main chapters III to VII show in detail the results of the main work packages explained above.

III. Monetary Valuation of Increased Mortality from Air Pollution

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1. Introduction

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

The report is structured in the following way. After a description of the research context in the introductory section, Section 2 provides an overview of the relevant literature including a summary of the unit values currently used in EU environmental decision-making. Section 3 describes the methodology used in the current study and the rationale for adopting an existing survey instrument. Section 4 presents the results from the individual country surveys and from a pooled analysis that utilizes data from all three countries. Section 4 also presents the discussion of the results and the consequent recommendations regarding unit values to be used in policy analysis. Section 5 concludes with an outline of outstanding issues and priorities for future research.

The impact-pathway approach to the estimation of environmental external costs adopted in the European Commission-funded ExternE Research Project requires – for its completion – the monetisation of the impact end-points identified by the modelling of pollution effects[1] arising from energy and transport fuel-cycles. In the case of air pollution, the epidemiological literature presented in previous phases of ExternE has signalled that exposure to a number of pollutants, including particulates, nitrates, sulphates and ozone, (e.g. European Commission, 1999), can lead to cases of immediate (acute) or delayed (chronic) premature death within a given population. There is therefore the need for a unit value to represent each estimated instance of premature death in the final estimation of environmental external costs.

The search for appropriate unit values has until now relied on the available literature. However, as explained in further detail below, the values that currently exist are generally not believed to express accurately the willingness-to-pay (WTP) that individuals might express, e.g. for the introduction of a new air quality regulation. More specifically, existing values are derived often in the context of the work-place (wage-risk studies) that estimate the willingness to accept (WTA) a higher wage rate in accordance with a greater risk of accidental death. Alternatively, attention has been given to the valuation of fatal transport accidents, the frequency of which might be expected to change with e.g. the introduction of new transport infrastructure.

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

- **The length of life-time lost on average through the impact**. Whereas the impact of premature death in the road or work context can be expected to be on

[1] See e.g. European Commission (1995) for details of the impact pathway methodology.

an individual of average age within the population and therefore result in the loss of about 35 years of life, air quality impacts are typically likely to lead to a loss of life of only a few weeks or months.

- **The state of health of the individual impacted**. Whilst the epidemiological literature suggests that air-pollution death is more likely to result in the case of an individual who has an already-existing impaired health condition, the typical victim of a premature death in the road or work context can be expected to be in good health.

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

- **Size of the risk change**. It has been suggested that the annual risk change associated with a realistic air pollution policy may be 10^{-4} whilst the risk valued in the transport accident context is typically 10^{-3}.

- **Context specificity**. The nature of the risk is perceived to be different according to the degree to which exposure to the risk is voluntary, the extent to which the potential impact is perceived to be controllable, and the size of the impact (in terms of number of deaths resulting). For example, premature death as a result of a road accident is likely to be perceived to be more voluntary to a death that results from ambient air pollution.

- **Immediacy of the impact**. Premature death resulting from a transport or workplace context is likely to result immediately following an accident. Conversely, there is often a lapse of time between being exposed to air pollution and feeling the health effects – that is, the effects are latent.

These differences give rise to the possibility that the unit values that should be applied to the air pollution external cost estimation differ from those derived in other contexts. To date the ExternE team has been constrained in adopting such values and then adjusting them to account for these differences, as far as theory and evidence allows. In practice, the main adaptation of the unit values derived from wage-risk (and other) studies has been to try to account for the length of life-time lost by changing the metric from the VSL (Value of Statistical Life), or VPF (Value of a Prevented Fatality), to the VOLY (Value of Life Years) – see Rabl (2003). Thus, in the ExternE transport project (Friedrich and Bickel (eds.) (2001)) the following explanation is given:

> The conceptual justification for this is that premature death matters because life is shortened and the amount of the shortening is material. The theoretical models that underlie the derivation of the WTP for a change in the risk of death are sensitive to the survival probabilities that the individual faces at the time the

valuation is made[2]. Hence *a priori* one would expect an empirical estimation of the WTP also to be sensitive to the amount by which life is shortened.

The basic assumption is made that the observed VOSL is the discounted present value of future years, allowing for the survival probabilities. In simple terms we assume the following to hold:

$$VSL_a(\{P\},r) = VLYL.\sum_{i=a}^{i=T} P_i(1+r)^{-i} \qquad (1)$$

Where a is the age of the person whose VSL has been estimated, P_i is the conditional probability of survival in year i, having survived to year i-1. T is the upper age bound and r is the discount rate. The above formula assumes that VOLY is independent of age (though it can straightforwardly be modified to allow for the possibility that VSL is age-dependent). This assumption will not in general be valid, but is made as a simplifying one that allows us to get an initial value for the kind of changes in survival probabilities that we expect to find in the area of air pollution.

The choice of WTP metric is discussed further in Section 2 below.

Outlining the differences in context from where the values are derived (wage risk, consumer markets etc.) and where they are used (air pollution), as we do in Section 2, below, indicates that there are reasonable grounds to expect that the unit values need not be the same. This provides the principal justification for the present study that tries to derive unit values that are more appropriate and reliable in policy use.

The need for reliability in policy analysis as a motivator for the current study is underscored when it is remembered that in previous ExternE analyses health impacts comprise 98% of the external costs from SO2 and 100% of those from particulates (European Commission (1999)), with mortality impacts accounting for at least 80% of these health impacts. Since this impact-pathway is critical to the scale of the external cost estimates it is important that the individual components of the pathway are as robust as possible.

This report presents the evidence from a survey-based (contingent valuation) study undertaken to address the types of issues highlighted above in the existing ExternE practice. As a consequence, there is an expectation that it will provide more reliable unit values to be used in policy analysis that uses the impact-pathway methodology.

[2] See, for example, M.J. Moore and K. Viscusi (1988), “The Quantity Adjusted Value of Life”, *Economic Inquiry*, (26), 368-388.

2. Literature Review

This section first outlines the principal methods used to date to measure unit values for premature death and highlights their appropriateness or not for measuring the welfare effects of a risk of loss of life due to air pollution. We then summarize current practice in policy appraisal that a unit value for premature death.

General Methodological Issues

Willingness to Pay (WTP) in the context of risks to life is defined as "the breakeven payment, per unit reduction in the probability of death, that leaves an individual's overall expected utility unchanged." (Shepard and Zeckhauser, 1982). In a more general context, the willingness to pay for an specific good or service is the sum of the amount of money individuals spend on the good or service plus the consumer surplus measure associated to the consumption of this good or service. Two general approaches have been used for the valuation of the benefits of lifesaving activities, including environmental programmes that reduce risks of death: the Human Capital approach and the Willingness to Pay approach (Cropper and Freeman (1991); Shepard and Zeckhauser (1982); Berger *et al.* (1994); Johansson (1995)). The first approach estimates measures the economic productivity of the individual whose life is at risk. It uses an individual's discounted lifetime earnings as its measure of value, assigning valuations in direct proportion to income. Alternatively, this approach assumes that the cost to society of a human death is the impact that such death has on national income or output, so that the value of a statistical life is measured in terms of its contribution to national income. This means that the value of preventing someone's death is equal to the gain in the present value of his or her future earnings. According to Kuchler and Golan (1999), the use of forgone earnings to measure the value of health and life depends on two assertions, that changes in health status are reflected in earnings and that national income is a reasonable measure of social welfare.

The Human Capital approach has the appeal of being easy to use but a number of ethical issues make it extremely contentious. For example, because of discounting and the time lag before children become productive participants in the labour market, the Human Capital approach places a much lower value on saving children's lives compared with saving lives of adults, who are in the labour force. Furthermore, because of earning differences among individuals of different gender and race, the Human Capital approach implicitly values saving the lives of women and nonwhites less than saving the lives of adult white males. Also, this approach assigns no value to retired or totally disabled people lives and does not account for the role of non-market production e.g. domestic housekeepers. Cropper and Freeman (1991) further argue that the most important criticism of the Human Capital approach is the inconsistency with the premises of welfare economics: it is each individual own preference that count for establishing the economic values used in cost-benefit analysis. These issues suggest that Human Capital measures are poor proxies for the willingness to pay measure for small changes in the risk of death. It does not reflect the probabilistic nature of death and individuals' different attitudes towards risks.

The Willingness to Pay approach has its basis in the assumption that changes in individuals' economic welfare can be valued according to what they are willing (and able) to pay to achieve that change. According to this assumption, individuals treat longevity like any consumption good and reveal their preferences through the choices that involve changes in the risk of death and other economic goods whose values can be measured in monetary terms. That is, in many situations individuals act as if their preference functions included life expectancy or the probability of death as arguments, and make a variety of choices that involve trading off changes in their risk of death for other economic goods. When what is being changed can be measured in monetary terms, the individual willingness to pay is revealed by these choices. The underlying assumption of WTP is that individuals are the best judges of their own welfare and are knowledgeable about the risks.

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

Compensating Wage Method

To date, the Compensating Wage method has been the predominant empirical approach to assess willingness to pay for risk reductions of premature death. The method uses labour market data on wage differentials for jobs with health risks and assumes that workers understand very well the workplace risk involved and that the additional wage workers receive when they undertake risky positions reflects risk choice. In other words, the Compensating Wage approach relies on the assumption that workers will accept exposure to some level of risk in return to some compensation. In general, it is estimated a hedonic wage function where wages are specified as a function of personal characteristics of the worker – income, age, sex, education, and health status - and the characteristics of the job. Among the latter, the fatality risk level of the job, benefits paid in case of injury on the job and benefits in the event of fatal accident can be cited as examples.

Compensating Wage models are consistent with the Willingness to Pay approach in the sense that they recognise that individuals have unique preferences over risky alternatives and that they have opportunities to reduce risks, depending on their labour skills. These models postulate that part of the differences in risk preferences are systematic and depend on objective and measurable individual characteristics. However,

> "Much of the criticism of the Compensating Wage approach centres on its assumptions concerning the labour market. Many critics argue that the actual labour market bears little resemblance to the labour market described in Compensating Wage models. The Compensating Wage approach assumes that workers are fully cognisant of the

> extent and consequences of the on-the-job risks they face, that labour market is strictly competitive, and that insurance markets are actuarially correct, with premiums and payouts matched to accurately assessed risks." (Kuchler and Golan, 1999).

Well known specific difficulties include:

- Omitted variables bias and endogeneity: failing to capture all of the determinants of a worker's wage in a hedonic wage equation may result in biased results if the unobserved variables are correlated with the observed variables, since dangerous jobs are often unpleasant in other respects. For example, one may find a correlation between injury risk and physical exertion required for a job or risk and environmental factors such as noise, heat, or odour. Various studies have demonstrated how omitting injury risk affects the estimation of mortality risk, indicating that a positive bias in the mortality risk measure is introduced when the wage equation omits injury risk.

  While including injury risk in a regression model could address concern about one omitted variable, other possible influences on wages that could be correlated with mortality risk may not be easily measured. For example, individuals may systematically differ in unobserved characteristics, which affect their productivity and earnings in dangerous jobs, and so these unobservable will affect their choice of job risk[3]. The studies reviewed by Viscusi and Aldy (2003) indicate that models that fail to account for heterogeneity in unobserved productivity may bias estimates of the risk premium by about 50%.

- Endogeneity: the issue here being that the dependent variable (wage) is explained by, among others, the risk variable, which simultaneously depends on wage, since "the level of risk that workers will be willing to undertake is negatively related to their wealth, assuming that safety is a normal good." Viscusi (1978). Gunderson and Hyatt (2001) empirically tested the alternative econometric models suggested by Viscusi (1978) and Garen (1988), identifying significant differences in the VSL estimates between the usual econometric model (OLS) and the proposed alternatives.

Empirical evidence

A recent study by Viscusi and Aldy (2003) reviews a large number of more recent wage-risk studies. The European studies – mostly from the UK – are summarized in Table 1 below.

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[3] Garen, J.E. (1988) "Compensating Wage Differentials and the Endogeneity of Job Riskiness", *Review of Economics and Statistics*, 73(4).

Table 1 Summary of European Labour Market Studies of the VSL

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

The range of values generated by these studies is a little disconcerting and reflect the different model specifications used. A conservative mean value of VSL from the lower end of these ranges is around €5 million.

A meta-analysis of 17 studies by CSERGE (1999) generated a range of VSL between €2.9 million and €100 million. The weighted (by sample size) arithmetic mean, when biases introduced by sample data and the analytical approach were controlled, was €6.5 million (2002 prices).

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

The Avertive Behaviour Method

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

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

The main criticism of the avertive behaviour method is that averting behaviours used in most studies, like wearing seatbelts or purchasing smoke detectors, are yes/no decisions, where the consumer decides or not to buy the averting good provided his or her marginal benefit is not less than the marginal cost of purchasing the good. The marginal cost equals the marginal benefit only for the last person to purchase the averting good, for all other consumers, the willingness to pay exceeds the marginal cost of a reduction in the conditional probability of death. However, it is possible to estimate the average willingness to pay using a probit or logit model of averting behaviour.

Another problem of the avertive behaviour method arises when the averting activity produces joint benefits, such as reducing the risk of injury or property damage as well as the risk of death. In practice, researchers deal with this problem either treating the value of joint products as zero, and than obtaining an upper bound to willingness to pay, or by assuming that the value of injury is some multiple of the value of a statistical life. Cropper and Freeman (1991) conclude that because of the problems cited above, especially the discreteness of the averting activity, the estimates of the value of a statistical life obtained from the averting behaviour method are lower than estimates obtained from other valuation methods.

Empirical Evidence

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

Contingent Valuation Method (CVM)

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

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

Table 2 provides a summary of the main biases that may be generated in a Contingent Valuation study.

The most serious problem related to Contingent Valuation studies may be the fact that the method provides hypothetical answers to hypothetical questions, i.e. no real payment is undertaken. This fact may induce the respondent to overlook his or her budget constraint, consequently overestimating his or her stated willingness to pay. In the context of risk and safety, the Contingent Valuation method involves asking members of a representative sample of the population at risk about their willingness to pay for a small hypothetical improvement in their safety. According to Beattie *et al.* (1998), people's *ex-ante* willingness to pay to reduce risk will tend to vary with their perceptions of the attitudes towards the characteristics of different hazards, such as the extent to which the hazard analysed is seen to be voluntarily assumed, under potential victims' own control, their own responsibility, well understood, and so on. The authors argue that there are evidences of apparent anomalies and inconsistencies in responses to willingness to pay questions in the safety and environmental fields. The most common inconsistencies involve embedding, scope and sequencing effects. The first two effects refer to the tendency of many Contingent Valuation respondents to report the same willingness to pay for a comprehensive bundle of safety or environmental good as for a proper subset of the bundle. Sequencing effects reflect a tendency for the order in which a sequence of Contingent Valuation questions are presented to respondents to have a significant impact on the willingness to pay responses.

The applicability of the contingent valuation method in the air pollution context appears to be high since the survey instrument allows the researcher to relate the WTP question precisely to the nature of the commodity to be valued – something that is not so easily possible in the market-based approaches. Its success therefore is determined by how effectively the survey instrument minimises the biases listed above. Most importantly, the scenario elements of the hypothetical market in the survey instrument must be understandable, meaningful and plausible to respondents.

Table 2 Typology of Potential Response Effect Biases in Contingent Valuation

1) *Incentives to misrepresent responses*
Biases in this class occur when a respondent misrepresents his or her true willingness to pay (WTP)
A *Strategic bias*: where a respondent gives a WTP amount that differs from his or her true WTP her true amount (conditional on the perceived information) in an attempt to influence the provision of Good and/or the respondent's level of payment for the good.
B *Compliance bias*
i *Sponsor bias*: where a respondent gives a WTP amount that differs from his or her true WTP amount in an attempt to comply with the presumed expectations of the (assumed) sponsor.
ii *Interviewer bias*: where a respondent gives a WTP amount that differs from his or her true WTP amount in an attempt to either please or gain status in the eyes of a particular interviewer.
2) *Implied value cues*
These biases occur when elements of the contingent market are treated by respondents as providing Information about the 'correct' value for the good.
A *Starting point bias*: where the elicitation method or payment vehicle directly or indirectly introduces a potential WTP amount that influences the WTP amount given by a respondent
B *Range bias*: where the elicitation method presents a range of potential WTP amounts that influences a respondent's WTP amount.
C *Relational bias*: where the description of the good presents information about its relationship to other public or private commodities that influences a respondent's WTP amount.
D *Importance bias*: where the act of being interviewed or some feature of the instrument suggests to The respondent that one or more levels of the amenity has value.
3) *Scenario misspecification*
Biases in this category occur when a respondent does not respond to the correct contingent scenario. Except in A, it is presumed that the intended scenario is correct and that the error occurs because the respondent does not understand the scenario as the researcher intends to be understood
A *Theoretical misspecification bias*: where the scenario specified by the researcher is incorrect in terms of economic theory or the major policy elements.
B *Amenity misspecification bias*: where the perceived good being valued differs from the intended one.
i *Symbolic*: where a respondent values a symbolic entity instead of the researcher's intended good.
ii *Part-whole*: where a respondent values a larger or a smaller entity than the researcher's intended Good.
a *Geographical part-whole*: where a respondent values a good whose spatial attributes are larger or smaller than the spatial attributes of the researcher's intended good.
b *Benefit part-whole*: where respondent includes a broader or a narrower range of benefits in Valuing a good than intended by the researcher.
c *Policy package part-whole*: where a respondent values a broader or narrower policy package than the one intended by the researcher.
iii *Metric:* where a respondent values the amenity on a different (and usually less precise) metric scale than the one intended by the researcher.
iv *Probability of provision*: where a respondent values a good whose probability of provision Differs from that intended by the researcher.
C *Context misspecification*: where the perceived context of the market differs from the intended context.
i *Payment vehicle*: where the payment vehicle is either misperceived or is itself valued in a way not intended by the researcher.
ii *Property right*: where the property right perceived for the good differs from that intended by the researcher.

iii *Method of provision*: where the intended method of provision is either misperceived or is itself Valued in a way not intended by the researcher.
iv *Budget constraint*: where the perceived budget constraint differs from the budget constraint the researcher intended to invoke.
v *Elicitation question*: where the perceived elicitation question fails to convey a request for a firm commitment to pay the highest amount the respondent will realistically pay before preferring to do without the amenity.
vi *Instrument context*: where the intended context or reference frame conveyed by the preliminary non-scenario material differs from that perceived by the respondent.
vii *Question order*: where a sequence of questions, which should not have an effect, does have an effect on a respondent's WTP amount, do in fact have an effect?

Source: Mitchell and Carson (1989) and Johansson (1995).

Empirical evidence

In this sub-section, we give a brief review of evidence based on CVM studies that relate to our search for unit values in the air pollution context, and in particular the issues of age, health status and context. A main reason for preferring not to rely on the VSLs generated by compensating wage studies is that the age of the victim is likely to be much younger in the work place context than in the air pollution context. The same is true of the road transport accident context, where a recent CVM study by (Carthy et al. (1999)), found a VSL of approximately €1 million.

The first study to address the issue of age dependency of VSLs was by Jones-Lee (1989) which examined individuals' WTP for reducing the risk of serious motor vehicle accidents. Based on a central VSL of €4 million at age 40, the age VSL variance was found to have an inverted U-shape – as shown in Table 3 below.

Table 3 Mean Estimates of VSL for Different Ages as a Percentage of VSL at Age 40

Age	20	25	30	35	40	45	50	55	60	65	70	75
VSL at age 40	68	79	88	95	100	103	104	102	99	94	86	77

Source: Jones-Lee (1989)

Other supporting evidence for a pattern of VSL declining with age is found in Desaigues and Rabl (1995) and Krupnick et al. (2000) – the latter using the survey instrument adopted in the present study in the Canadian context.

A more recent study is that of Johannesson and Johansson (1996) who use the contingent valuation method to look at the WTP of different respondents, aged 18-69, for a device that will increase life expectancy by one year at age 75. A sample of the results obtained is reported in Table 4.

Table 4 WTP (EURO, 2002) for 1 Year of Life at Age 75 and Corresponding Values for 1 Year of Life Immediately

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

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

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

Regarding a relationship between *health status* and VSL, the CVM evidence is very limited and inconclusive. The principal studies that have explored this linkage are Johannesson and Johansson (1997) who found that WTP values declined with poorer health status, whilst Rudnick (2000) found no significant evidence of a relationship.

The relationship between WTP and *context* is similarly under-developed in terms of primary CVM studies. The main studies, by Jones-Lee and Loomes (1994, 1995, 1996) and Covey et al (1995), reported in Rowlatt et al (1998) consider the road transport accident VSL in relation to those for Underground rail accident risks, food risks, risks to third parties living in the vicinity of major airports and domestic fire risks. The perceived involuntariness of the underground rail risk attracted a 50% premium on the road VSL, whilst a 25% discount is attached to the risk of a domestic fire. The latter result was thought to reflect the high degree of voluntariness or controllability in this context. No evidence was found to support an adjustment to the road accident VSL for scale of the accident (i.e. in the case of the underground accident or residents proximity to airports contexts). Thus, the limited evidence suggests context relating to voluntariness is likely to be important in determining WTP but the weight of evidence for this is not yet strong enough to draw this as a strong conclusion.

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

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

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

Alternative Metrics

There has been considerable debate within the ExternE team as to whether the Value of Statistical Life (VSL) should be placed by the Value of Life Years (VOLY) as the principal metric by which to value incidence of premature death from air pollution. Table 5 below summarizes some of this thinking. A key argument in this debate has been proposed by Rabl (2002). He shows that the number of deaths that can be attributed to this cause is only observable in mortality statistics when the exposure-death effect is sufficiently instantaneous that the initial increase in death rate is not obscured by the subsequent depletion of the population who would otherwise die later.

Rabl argues that the usual case is that the impact of air pollution is not instantaneous but the cumulative result after years of exposure, so that the number of deaths is not observable[4]. As a result, it is impossible to tell whether a given exposure has resulted in a small number of people losing a large amount of life expectancy or a lot of people losing a small amount of life expectancy. In this case only the average number of years of life lost are calculable and so makes a strong case for the use of VOLYs in the context of air pollution.

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

Table 5 Appropriateness of Value Metrics in different Contexts

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

[1] Theory and empirical evidence support an inverted U - shape but theory excludes value of survival and possibilities of changes in preferences for risk as we grow older. Moreover, empirical evidence is quite limited.

[2] Theory might suggest declining values with age (loss of life expectancy falls as you get older). But we still must allow for changes to attitudes to risk etc.

Appendix 1 outlines the current practice followed in policy applications of WTP to avoid premature death. The most detailed guidance in Europe is provided by the European Commission itself and the UK, and we summarize the policy values that are currently used in these countries in Table 6.

Table 6 Current policy guidance on unit values

Adjustment factor	EC Guideline	UK Govt. Guideline
Baseline VSL	Central: €1.4 million Range: €0.65 - €3.5 million	Central: €1.2 million
Context	50% premium for cancer	Involuntariness – multiply by 2
Age	Multiplier of 0.7 (applies to central value only)	Multiplier of 0.7
Health	No adjustment	Upper estimate: no adjustment. L.E adjustment – multiply by 0.007 or 0.08 Quality of life adjustment – multiply by 0.28 or 0.92
Cultural	No adjustment	No adjustment
Income	No adjustment	No adjustment
Final Unit Values	Central: €1 million Range: €0.65 - €3.5 million	Central: €0.134 million Range: €0.0029 - €1.75 million
Futurity	Discount rate: 4%	Discount rate: 1%

3. Justification of Research Methodology

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

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

In the following paragraphs we outline the structure of the survey instrument and rehearse key arguments relating to important design features, including the ways in which it attempts to address a number of biases associated with contingent valuation studies.

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

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

Description of the survey

The components of the questionnaire are described in this section whilst a number of the key screens from the computerized survey instrument are presented in Annex 4.

Component 1 Introduction to the survey, and reassurance that it is not a marketing exercise but that the respondents' opinions are being sought. The respondent's age and gender are requested since the remainder of the survey is affected by the answers to these questions.

Component 2 Establishment of health status, in which the health of relatives and the individual are recorded, focusing on the presence or absence of various chronic diseases. This has several purposes. The questions are straightforward and therefore help to get the respondent used to the screens; they encourage the respondent to think about their health *before* responding to the WTP questions. Being few in number, these questions do not encumber the survey.

The respondent is asked to rate their current health relative to others of their age and gender, and to rate their expected health in ten years relative to their health today. They are also asked to rate their expected health at age 70 relative to their expected health in 10 years. These questions are relevant because the WTP questions are for mortality risk reductions over the next 10 years and from age 70 to 80. Perceived life expectancy is requested and is used to establish whether those with a longer perceived life expectancy would be willing to pay more for a future risk reduction than others.

Component 3 This component educates the respondent about probabilities in general and specifically about risks of death. The main purpose of this section is to communicate facts about probabilities clearly and test for comprehension, eschewing tests of mathematical ability. Screens move from simple coin flips to a roll of the die and then introduce the idea of a grid, the total number of squares representing possible outcomes, and red squares representing outcomes of a particular type. A key graphic – 1,000 grid squares, with several coloured red, represents the risk of death.

The expression of probabilities as X per 1,000 is the basic unit of risk communication in the survey. This unit was chosen following extensive testing in North America. It was concluded that the use of grids with more than 1,000 squares (i.e. 10,000 or 100,000) results in reduced cognition and a tendency to ignore small risk changes as being insignificant. Because annual risk changes associated with air pollution policy are smaller than 1 in 1,000, however, the commodity is expressed as a risk change over 10 years *totalling* x per 1,000. Baseline risks and payment schedules are also put in 10-year terms. The grid shows red squares dispersed, to indicate the randomness of risks.

The rationale for mortality being discussed in 10-year intervals was that focus groups and pre-testing in North America showed that respondents find it considerably easier to conceptualize the possibility of dying in a 10-year period than over a one-year period. The use of 10-year intervals allows us to represent risks in terms of chances per 1,000, which can be shown easily on the grid. In addition, the one-year risk change is implicitly approximately 1/10,000, which is in the appropriate range for capturing the risk reductions associated with pollution reductions. The use of this mechanism is central to the strategy to reduce potential scoping problems.

Understanding of the concept of risk is tested by first describing two people, Person 1 and Person 2. These people are identical in every way, except one has a 5 in 1,000 chance of dying over the next 10 years while person 2 has a 10 in 1000 chance of dying over the next 10 years. The respondent is shown side-by-side graphs of the risks for these people and asked to pick which person has the largest chance of dying. Respondents who cannot answer this question correctly will not be able to perform on the survey and these are therefore not included in the subsequent analysis. Even if a respondent can distinguish these risks, he or she may not feel that the difference in risk is "significant." To identify such respondents, it is asked which of these two people they would rather be (including "indifferent" as a possible answer). A wrong answer or "indifference" would be hypothesized to result in lower WTP than for those providing the right answer.

Component 4 This component provides baseline risks, using the respondent's age and gender information, and additional information about these risks to put them into context. The idea of baseline risks is introduced by showing the effect of age on baseline risks in ten-year increments, both verbally and with a graph. The respondent sees a grid with the appropriate number of red squares representing the 10-year baseline risks for someone of their age and gender. To help fix this baseline in the respondent's mind, he or she is asked to create his or her own baseline risk graph by pushing a key. This procedure, along with other specific features within the study, is intended to ensure that *hypothetical bias* is reduced.

Component 5. One difficulty in asking people to value quantitative risk reductions is that, although people often engage in risk-reducing behaviour (e.g., cancer screening tests, taking medication to reduce their blood pressure or cholesterol levels), they may have no idea how much these actions reduce their risk of dying or their true costs. Information is therefore presented to the respondents on age- and gender-specific leading causes of death and common risk-mitigating behaviour – both medical and non-medical. Illustrative risk reductions for these are provided (estimated from the literature) along with cost ratings. The idea is also introduced that even though a procedure or action may be free to the insured, someone still pays.

The cost ratings are used for several reasons. First, actual cost estimates are problematic because they might anchor later WTP responses. Second, actual cost estimates might introduce dissonance between the costs the respondent actually pays and social costs - the latter usually being higher. Third, actual costs are not needed, because the purpose of this section is only to leave respondents with the knowledge that in every day life, they do pay small amounts of money to reduce mortality risks by a fractional amount.

Component 6. This component seeks to elicit WTP for risk reductions of a given magnitude, occurring at a specified time, using dichotomous choice methods with one follow-up. The dichotomous choice elicitation method is that recommended by the NOAA guidelines. The justification for adopting this method given by the original designers of this instrument is that it reduces the possibility of strategic bias. Follow-up questions are used because they allow the econometrician to dramatically improve the statistical efficiency of the WTP estimates obtained from the study.

An example of the WTP questions is:

> *Suppose that a new product becomes available that, when used over the next ten years, would reduce your chance of dying from a disease or illness. This product would reduce your total chance of dying over the next ten years from X to Y.*
>
> *If you were to take this product you would have to pay the full amount of the cost out of your own pocket each year for the next ten years. For the product to have its full effect, you would need to use it every year for all ten years.*
>
> *We realize that most people will not simply accept the idea that this product is guaranteed to work without some proof. In answering the next questions, please assume that the product has been demonstrated to be safe and effective in tests required by the UK Government.*
>
> *Keeping in mind that you would have less money to spend on other things, would you be willing to pay €Z per year (10 times Z total) to purchase this product?*

The North American team believed that there were compelling reasons for keeping the agent for the risk reduction and the payment vehicle completely “abstract”, as in this example. Whilst this departs from the NOAA panel recommendations, it was felt that there was sufficient evidence (see e.g. Hurd and McGarry (1997), and Cropper *et al* (1994)) to show that respondents are willing and able to make choices among abstract life-saving programs allowing respondents to focus on the size of the risk reduction itself and the effect it has on oneself, thereby avoiding various potential biases. Moreover, making the risks specific may result in reduced values since people may not believe that specific risks apply to them. In the specific case of reductions in air pollution, there are numerous non-health benefits, and benefits to others, which people may or may not factor into their valuation. It was argued that these factors may lead to distorted estimates of the value to the individual of the health benefits.

In addition, the means by which risk is reduced is presented as a private good rather than through public spending. This was because public spending is conceived by respondents as benefiting people in general whereas, as pointed out by Jones-Lee (1991), the appropriate welfare measure is what people would be willing to pay to reduce risk to themselves.

WTP per year over the next 10-year period is asked for a risk change of 5/1,000 over the same period, and 1 in 1,000 over the same 10-year period. The 10-year sum of the annual payments is also provided. For the third WTP question (asked only of individuals 60 or less), the respondent is then told his or her gender-specific chance of dying between ages 70 and 80 and is asked, through dichotomous choice questions, their WTP each year over the next ten years for a future risk reduction beginning at age 70 and ending at age 80 *which totals* 5 in 1,000. The respondent is reminded that there is a chance he or she may not survive to age 70, making a payment today useless. He or she is then given the opportunity to revise their bid. During an extensive debriefing section of the survey, the respondent is asked whether they thought about their health state during this future period. Each WTP question is followed by a screen to gauge the strength of a respondent's conviction in his WTP responses. The North American experience has shown that the variance of WTP is smaller for the sample who have strong convictions.

Components 3-6 aim to ensure that whilst the respondent is given a rigorous understanding of the notion of risk the information requirements required in explaining a specific cause of the increased risk (air pollution) and the policies needed to reduce the risk are minimised if the risk reducing agent and payment vehicle are left abstract. The intention is to minimise information bias.

Component 7. This includes debriefing questions. Each debriefing question probes the state of the respondent’s mind when they answered the various WTP questions and some other questions. Answers to these questions are to be used to explain variation in WTP. For instance, if the respondent felt it was unreasonable to ask for payment today to reduce risks in the future, we would expect WTP from this respondent to be less than someone who felt this was reasonable. Similarly, respondents who were thinking about morbidity improvements as a result of the product, as well as mortality improvements, would be expected to be WTP more than those who only thought about mortality risk reductions. The short form SF36 on health status is also included in the survey. Finally, the respondent is given the opportunity to review the values (s)he has chosen and amend, if (s)he so wishes.

4. Results of Country Studies and Pooled Analysis

In Appendix 3 to this report we present the results from the individual country surveys in UK, France and Italy. Here, we summarize the individual country studies and present the results from an econometric analysis that pools the data from the individual surveys. The latter analysis allows us to explore the possibility that unit values for the EU as a whole can be based on the survey data from a range of countries. Alternatively it allows us to speculate as to whether unit values in individual countries can be explained by observable variables e.g. income, or whether cultural differences render any such analysis and derivation of common unit values a fruitless exercise.

Willingness to Pay for Mortality Risk Reductions: Preliminary Results from Europe: Pooled Analysis

4.1. Introduction

The purpose of this document is to summarize the results of a contingent valuation survey eliciting willingness to pay (WTP) for reductions in one's own risk of death. The survey was self-administered using the computer to samples of respondents in three countries — the UK, Italy, and France, following the protocol developed by Krupnick et al. (2001). The questionnaire had previously been administered to a sample of Canadians and a sample of US residents. Results from these surveys are presented in Krupnick et al. (2001) and Alberini *et al.* (forthcoming).

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

In this report, attention is restricted to WTP for the 5 in 1000 risk reduction over the next 10 years. Future updates to this report will examine WTP for the future risk reduction, as well as alternative econometric specifications for WTP for the current risk reduction.

The remainder of this report is organized as follows. In section 4.2, we summarize sampling procedures and experimental designs, comparing them with those of two previous rounds of the survey, which took place in Canada and the US. In section 4.3, we present descriptive statistics for the respondents. In section 4.4, we examine the respondents' comprehension of risks. In section 4.5, we report descriptive statistics about the respondent's health status. In section 4.6, we present WTP figures, and in 4.7 regression models that test internal validity of the responses to the payment questions. The latter are based on pooling the data from the UK, Italy, and France. Section 4.8 summarizes the recommendations gathered from this study: values that are recommended to be used for the calculation of new results (Chapter VII).

4.2. Mode of Administration and Sampling

In Canada, respondents were first contacted among the residents of Hamilton, Ontario, using random digit dialing, and were asked to report to a centralized facility in Hamilton. In the US, the questionnaire was administered to a sample selected from Knowledge Networks' Web-TV panel. In the UK and France, respondents were contacted in the Bath and Strasbourg area using a mix of random digit dialing, in-street intercept, and snowballing, whereby one respondent is asked to submit names of acquaintances. In Italy, respondents were selected among participants in computer classes at the FEEM's Multimedia Library in Venice, Milan, Turin and Genoa, and from workers of the Milan area.

In Italy and the UK, the risk reductions to be valued by the respondents were those used in Wave 1 of the Canada and US studies. Specifically, people were asked to value a 5 in 1000 risk reduction, a 1 in 1000 risk reduction, and a reduction of 5 in 1000 to be experienced at age 70. The France study also implemented the Wave 2 design, whereby the 1 in 1000 risk reduction was valued first. Table 7 also reports the sample sizes, which are of the order of about 300 in the three European countries.

Table 7 Sample size and experiment design for the five-country study

	Canada	US (national survey)	UK	Italy	France
N	930	1200	330	292	299
Locale of the Study	Hamilton, Ontario	Nation-wide survey	Bath	Venice, Genoa, Milan and Turin	Strasbourg
Experimental Design	Wave 1 and wave 2	Wave 1 and wave 2	Wave 1	Wave 1	Wave 1 and wave 2

4.3. Descriptive Statistics of the Respondents

The sampling plan restricted attention to persons older than 40 years of age and specified the proportions of the samples for the various age groups. The average age in the three European countries ranges from 55 to 58, as is appropriate and consistent with the sampling frame.

The samples are relatively well balanced in terms of gender, with only a slight prevalence of women over men, and the average number of years of schooling ranges from 11 (for the French study) to about 14 (for the UK).

Mean and median annual household incomes are reported in the original currency, in euro, and in PPP US$. To convert GBP to euro, we multiplied the GBP amounts by 1.46. To convert euro to US, we multiplied the Italian figures by 0.813, the French figures by 0.917, and the UK figures by 0.918.

Table 8 Descriptive Statistics of the Respondents

	UK	Italy	France
Age	58	57	55
Age group 40-49	20%	28%	33%
Age group 50-59	34%	33%	29%
Age group 60-69	33%	23%	26%
Age group 70 and older	11%	14%	10%
Male	49%	48%	47%
Income Mean Median	GBP 27,463 26,500	Euro 40,115 25,000	French Francs 211,144 210,000
Income in EUR Mean Median	 40,096 38,690	 40,115 25,000	 32,186 32,012
Income in 2002 US $ using PPP: Mean Median	 36,768 35,478	 32,613 20,325	 29,571 29,411
Education (years of schooling)	14	13	11

4.4. Baseline Risks and Health Status.

Table 9 reports the health status of the respondents, based on their answers to questions about cardiovascular and respiratory problems. It also reports the baseline risk for each respondent, which is based on published statistics and depends on the respondent's age and gender.

Table 9 Health status of the Respondents

	UK	Italy	France
Rates own health as good or excellent relative to others same age	61 percent	39 percent	39 percent
CARDIO	8 percent	12 percent	12 percent
LUNGS	15 percent	12 percent	14 percent
PRESSURE (high blood pressure)	28 percent	21 percent	21 percent
CANC	6 percent	6 percent	7 percent
Any one of CARDIO, LUNGS, PRESSURE, or a stroke (cancer excluded)	43 percent	39 percent	45 percent
Baseline risk of dying over the next 10 years	199	50	109

4.5. Risk Comprehension and Acceptance of the Survey

Table 10 displays the percentages of respondents who failed the probability test and choice questions or otherwise report having problem understanding the concept of risk.

Table 10 Percent of the sample who have various problems with risk comprehension. Based on complete samples.

	UK	Italy	France
Wrong answer to the probability test question	15	12	23
Confirms wrong answer to the probability test question	0.91	3	4
Probability choice question: -- Wrong answer -- indifferent	 14 7	 12 11	 10 22
Confirms wrong answer in the probability choice question	1.52	3.08	1.34
Thinks he/she understands probabilities poorly (FLAG6=1)	27	27	*
FLAG1=1	2.5	3.8	2

* In France, all respondents answered a 5 or less to this question. It is not clear at this time whether the respondents were only shown 5 response categories, or whether they spontaneously chose the 1-5 answers.

4.6. Responses to the Payment Questions and WTP Figures

In Figure 1, we show the percentage of ‘yes’ responses to the initial payment questions for the 5 in 1000 risk reduction. Economic theory suggests that the percentage of ‘yes’ responses should decline with the bid amount, and indeed this expectation in borne out in the data. It should also be noted that only in the UK sample the respondents were offered bid amounts that are greater than median WTP. In the France and Italy samples, bid amount were always less than or just about equal to median WTP. This may have repercussion in our estimation of mean and median WTP, since in previous research (Alberini and Longo, draft paper) it is shown that with skewed distributions of WTP it is important to identify the upper tail of the distribution to obtain a reliable estimate of mean WTP. Table 11 shows the initial bid values that the respondents were presented with.

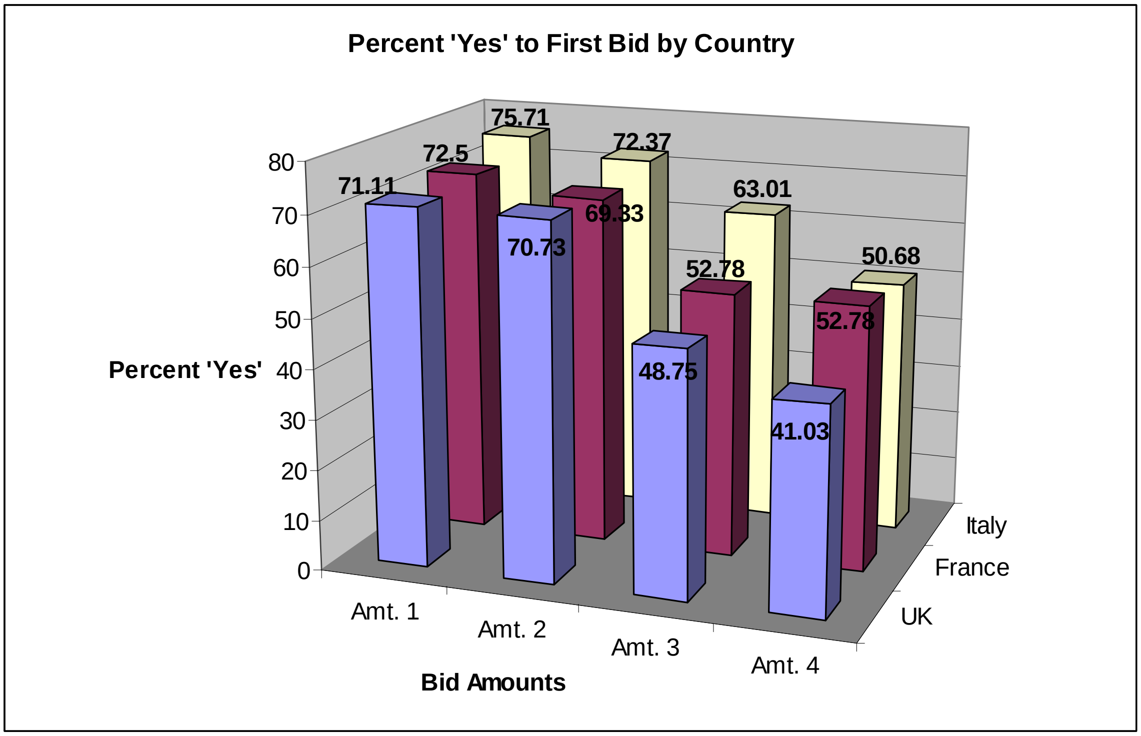

Figure 1 Percentage of responses with "yes" to question of initial payment for a risk reduction of 5 in 1000

Table 11 Bid design by country.

	Initial bid	If yes	If no
Canada (Canadian dollars)	100	225	50
	225	750	100
	750	1100	225
	1100	1500	750
US (US dollars)	70	150	30
	150	500	70
	500	725	150
	725		500
UK (Pound Sterling)	45	100	20
	100	325	45
	325	475	100
	475	650	325
Italy (Euro)	80	170	35
	170	570	80
	570	830	170
	830	1140	570
France (Francs)	500	200	1000
	1000	500	3500
	3500	1000	5000
	5000	3500	7000

To obtain estimates of mean and median WTP, we combine the responses to the initial and follow-up payment questions to form intervals around the respondent's (unobserved) WTP amount. For example, if a respondent is willing to pay the initial bid of, say, €100, and declines to pay the follow-up amount of €225, it is assumed that his WTP falls between €100 and €225. We further assume that WTP follows the Weibull distribution with scale parameter σ and shape θ, and estimate these parameters using the method of maximum likelihood. The log likelihood function of the WTP data is:

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

where WTP^L and WTP^U are the lower and upper bound of the interval around the respondent's WTP amount. Equation (1) describes an interval-data model. We first fit this model separately for the Italy, France and UK data, and in the next section we consider pooled-data models.

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

Another reason for preferring the Weibull distribution is that in our experience the Weibull has proven generally better-behaved than the other positively skewed distributions (like the lognormal). The Weibull and the other distributions generally agree in terms of their estimates of median WTP, but may produce very different figures for mean WTP. In addition, the Weibull distribution has a flexible shape: Depending on the value of the shape parameter theta, the density of the Weibull variate can be positively skewed (for theta between 0 and 3.6), symmetric (for theta approximately equal to 3.6), and even negatively skewed (for theta greater than 3.6).

The mean of a Weibull variate is equal to:

$$(2) \qquad \sigma \cdot \Gamma\left(\frac{1}{\theta} + 1\right)$$

while median WTP is equal to:

$$(3) \qquad \sigma \cdot \left[-\ln(0.5)\right]^{1/\theta}.$$

With WTP, experience suggests that mean WTP tends to be two or even three times as large as median WTP. We regard median WTP as a conservative, but robust and more reliable, estimate. For this reason, we report median WTP figures for the 5 in 1000 risk reduction in Table 12 below.

Table 12 Median WTP for the 5 in 1000 risk reduction beginning now. Wave 1, Double-bounded Weibull model. Uncleaned samples. Annual WTP**.**

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

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

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

4.7. Pooled-data Models and Internal Validity Tests

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

We pool the data from the three European countries to increase the sample size and to be able to provide recommendation for VSL figures to use for EC policy purposes. The first specification of this econometric model (column (A) of Table 12) includes an intercept and an income covariate. It can be regarded as the pooled-data equivalent of the models used to produce the estimates of mean and median WTP of Table 11. The income variable is included in an effort to answer the question whether WTP for the 5 in 1000 risk reduction and the VSL should be allowed to be depend on a country's income.

In column (B) we include country dummy variables in order to test whether there are country-specific factors that are influencing WTP additional to the other explanatory variables.

In column (C) we include age dummies, gender, education, and measures of the health status of the respondent. Specification (C) allow us to check whether the VSL should be adjusted for the beneficiary's age and health status in environmental policy applications. It should be noted that the sign of the age and health status variables is not known a priori. One would expect WTP to increase with baseline risk, but higher baseline risk implies lower remaining life, an offsetting effect if the value of each remaining life year is assumed to be constant. Under restrictive assumptions, Shepherd and Zeckhauser obtain an inverted-U shaped relationship between WTP and age. Similar considerations hold for the health status dummies.

One would expect, however, income to be positively correlated with WTP. The sign of education is not known a priori: someone with better understanding could give a lower or a higher WTP. In column (D), the regression is re-run with country dummies included among the covariates.

Table 13 Pooled data interval-data regressions for WTP. 5 in 1000 risk reduction.

	(A)	(B)	(C)	(D)
Intercept	6.4648** (0.126)	6.0057** (0.148)	6.7208** (0.342)	5.8024** (0.386)
Household income (thou. Euro)	0.0089** (0.0029)	0.0097** (0.0029)	0.0098** (0.0031)	0.0098** (0.0031)
Age 50-59 (dummy)			-0.0702 (0.196)	0.0245 (0.190)
Age 60-69 (dummy)			0.0391 (0.207)	0.2056 (0.204)
Age 70 or older (dummy)			-0.2144 (0.263)	-0.0748 (0.256)
Male (dummy)			-0.1831 (0.147)	-0.1842 (0.142)
Education			-0.0217 (0.023)	0.0072 (0.024)
Chronic respiratory or cardiovascular illness (dummy)			0.0409 (0.157)	0.076 (0.152)
ER or emergency room visit (dummy)			0.7445** (0.292)	0.5944* (0.282)
Has or had had cancer (dummy)			0.4399 (0.326)	0.4397 (0.315)
France dummy		0.8405** (0.205)		0.8636** (0.214)
Italy dummy		0.6556** (0.160)		0.6705** (0.162)
Shape parameter (θ)	0.7014 (0.042)	0.7276 (0.043)		0.7400 (0.044)

The results shown in column (A) imply that mean WTP for the 5 in 1000 risk reduction from the three European countries is €1129 per year (s.e. €132.5), while median WTP per year is pegged at €526 (s.e. €39.5). The implied VSLs are €2.258 million and €1.052, respectively.

Column (A) shows that income is significantly associated with WTP, a result that is consistent with expectations. The model implies that to predict median WTP for a country with income equal to Y thousand €, the following formula should be used:

$$(4)\ \exp(6.4648 + 0.0089 \cdot Y) \times [-\ln(0.5)]^{1.42}$$

Accordingly, a country with income equal to €20,000 should have an annual median WTP of €456. If Y=€15,000, median WTP is €435, and if Y=€27,000, median WTP is €484.

Column (B) includes country dummy variables to account for the different sampling frames at the different locales where the survey was administered. Holding household income the same, the French and the Italian respondents hold WTP values that are greater than their UK counterparts. In this specification, the coefficient of income is larger in magnitude than, but is within 10% of, its counterpart in specification (A).

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

4.8. Recommended values

Interpretation for VOLY

The discussion of the appropriate WTP metric for the air pollution context, summarized in Section 2 above, concluded that the epidemiological evidence dictated that the VOLY be adopted. Since we do not have direct estimates of VOLY – our survey generates VSLs – we rely upon a conversion relationship between changes in probabilities of death and changes to life expectancy. This relationship is established in Rabl (2002), which presents the equivalent change in life expectancy associated with the 5 in 1000 change in risk of premature death for different ages and sex, based on EU population statistics. It suggests, for example, that a person of age 55 will gain an equivalent of 40 days from a 5 in 1000 change in risk.

Recommended values for premature death in ExternE (NewExt)

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

2) To use to value air pollution impacts within ExternE we need to convert the WTP for 5: 1000 immediate risk change into a value of a life year (lost or gained). Rabl (2003) derives the changes in remaining life expectancy associated with the 5 in 1000 risk change over the next 10 years valued in this study, based on empirical life-tables[5]. According to Rabl's calculations, the extension in life expectancy ranges from 0.64 to 2.02 months, depending on the person's age and gender, and averages 1.23 months (37 days) for our sample. To find out the value of a life-expectancy extension of a month, we divide a respondent's WTP by that respondent's life expectancy extension. A Weibull double-bounded model pegs mean WTP at €1052 (s.e. 128.4) per year for each month of additional life expectancy. Median WTP is €465 (s.e. 33.3) for a month of life expectancy gains. Because in our survey the payments would be made every year for ten years, the total WTP figures for a life expectancy gain of one month are €10,520 and €4650 respectively. The implied values of a statistical life-year (VOLY) are €125,250

[5] A change in the probability of surviving the next 10 years changes the probabilities of surviving all future periods, conditional on being alive today. The sum of these future probabilities of surviving is a person's remaining lifetime. Rabl's calculations are based on an exponential hazard function, $h(t)=\alpha*\exp(\beta t)$, where t is current age, and α and β are equal to 5.09*E-5 and 0.093 for European Union males, respectively, and 1.72E-5 and 0.101, respectively, for European Union females.

and €55,800, respectively. Given the uncertainties, this might safely be rounded to €50,000.

3) The VOLY of € 50,000 is derived from an annual payment made over a ten-year period and as such does not require further discounting since we assume that the respondents have implicitly done this when giving their answer. Since available empirical evidence suggests that a typical time period of latency to elapse in the case of chronic air pollution-induced mortality is 5-7 years we may adopt this value for chronic mortality impacts, whilst noting that the life years lost (gained) after the time of death are not accounted for in this unit value. If, however, we assume that the VOLY of €50,000 is equivalent to the VOLY derived from life-table analysis, (following Hurley and Miller, (2004), and Friedrich and Bickel (eds) p92, (2001)), discounted at 3%, then the equivalent undiscounted VOLY is (50,000/0.67) = €74,627[6]. For calculating new results, this value is rounded to €75,000. This can be interpreted as a value for acute mortality as long as it is assumed that no other factors (e.g. a victim's health condition at time of death) affect WTP for these end-points.

4) Upper and lower bounds are estimated in the following way:

    a. The upper bound value is taken as that resulting from the results from the 1:1000 immediate risk change. We do not have pooled data for this risk change but instead use the UK results. These give a VSL of €3,310,000 and a VOLY (discounted) of €151,110. The corresponding undiscounted VOLY amounts to €225,000 (rounded).

    b. The lower bound estimate is derived from the results of the French questionnaire that uses a direct estimate of an equivalent change of life expectancy of €200. This converts to a VOLY of €18,250. The corresponding undiscounted VOLY amounts to €27,240.

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

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

5. Outstanding Issues and Future Work

The preceding sections of this report have outlined how the EC NewExt project has made progress in the valuation of premature death resulting from air pollution. Sections 1 and 2 reminded us of current evidence and current practice relating to this valuation objective. It was demonstrated that whilst the context of air pollution might suggest that direct transfers of other contexts is not appropriate, this is the only procedure possible given the lack of valuation studies in this context. It was also highlighted that the epidemiological evidence suggests that the appropriate metric is the value of life expectancy lost rather than the value of statistical life, on which almost all empirical valuation studies focus.

In order to fill this gap the project team committed to undertake a contingent valuation study in three European countries – France, UK and Italy. The only developed survey instrument designed specifically to address the valuation of death in the air pollution context was that of Alan Krupnick and colleagues from Resources For the Future (RFF) in the US, and as a sub-contractor to the project team, the project was able to adopt this same survey instrument. The detail of the survey is presented in Section 3 above. As well as benefiting from the RFF's experience of administering the survey in North America, the project significantly reduced the development costs associated with the construction of such an instrument. Nevertheless, the country teams conducted a series of focus groups and/or one-to-one testing in order to better understand how the respondents interpret the questionnaire.

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

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

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

The pooled results of the country studies are presented in Section 4; detailed country results can be found in the Appendices. The parametric analysis of the pooled data does not suggest that the VSL has a significant relationship with the age of the individual; but, this can differ in different countries. In the UK and Italy the econometric results of the pooled data do not show any significant relationship between health and VSL; this is not the case for France. The VSLs show some differences between the three countries but in the context of range of VSL in the literature, these differences are not that large. Using the Weibull regression estimation technique, the VSL is €772,000 for the UK, €1,448,000 for Italy, and €959,000 for France. When the data from the three country studies are pooled, a VSL of €1.052m is derived, and this value might safely be rounded to €1 million. A VOLY was then estimated by converting the WTP for the risk change, (5:1000), to an equivalent change in life expectancy (40 days), and multiplying up to a give a value for a life-year of €55,800. Given the uncertainties, this might safely be rounded to €50,000.

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

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

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

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

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

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

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Appendix 1 Policy Applications – Current Practice

This section outlines the current practice followed in policy applications of WTP to avoid premature death. The most detailed guidance in Europe is provided by the European Commission itself and the UK, and this section therefore focuses on these practices.

European Commission (EC)

The practice followed by the EC was developed on the basis of a meeting of valuation practitioners convened by DG Environment in November 2000[7]. The following structure to the valuation of premature death from air pollution was reached.

Baseline values.
A range of baseline VSLs were chosen to reflect the existing spread and uncertainty in the empirical literature, The 'best' estimate was to be treated as the central estimate with the 'upper' and 'lower' figures used for sensitivity analysis.

1) **Upper Limit** – the current **ExternE value** of around €3.5m (2000 prices), constructed on the basis of an informal meta-analysis of compensating wage, CVM and consumer market studies. Over-reliance on compensating wage studies was felt to be a weakness with this value.

2) **Best Estimate** - The UK Department of the Environment, Transport and the Regions' figure of €1.4m (2000 prices) for VSL in the transport accident context was thought to offer a strong starting point, being principally based upon a number of consistent Contingent Valuation Method (CVM) studies.

3) **Lower Estimate** - a value of €0.65m (2000 prices) for older people valuing risk, derived from the Krupnick et al application of the present survey instrument in Canada.

Adjustments for Context

Given that the best estimate is being transferred from a transport context, it was thought necessary to adjust it for the most important contextual factors[8]: age, health status, income, cultural differences and altruistic concerns. The headings below summaries the rationale for any subsequent adjustments made.

<u>Age</u>
As noted above, mortality incidents due to poor environmental quality tend to be concentrated amongst older people with a lower life expectancy. There are strong

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[7] See the full report of the workshop at:
europa.eu.int/comm/environment/enveco/others/proceedings_of_the_workshop.pdf
[8] The following discussion applies to the best and upper estimates, but only partially to the lower estimate since its original context is environmental

theoretical and empirical grounds for believing that the value for preventing a fatality declines with age. An adjustment of 0.7 from the central reference value from its transport context to a person aged 70 on the basis of the Krupnick et al study results.

Health

The hypothesis that people in a poor state of health will be less willing to pay to reduce risk because of the lower quality of the life they would be foregoing was judged not to be sufficiently supported by existing empirical evidence. No adjustment was therefore made on this basis.

Income

The recommended values are designed for application to the population of the EU. However, willingness (and ability) to pay for reductions in risk may vary with income. There is therefore a question as to whether these values should be adjusted for the income of the population at risk in the EU. However, since EU Member States do not discriminate within their own populations on the basis of income it was not thought appropriate for policy at the EU to do so either. In addition there was not thought sufficient empirical evidence to support the hypothesis.

Futurity

For chronic or latent effects associated with air pollution, it was thought appropriate to adopt the standard DG Environment discount rate of 4% for discounting these future impacts. It was agreed that sensitivity analysis should be carried out using a value of 2%.

Cultural differences

No adjustments were suggested to reflect significant cultural differences in preferences between populations since no evidence supported this adjustment.

Context

On the cause of death, and cancer in particular, there is little evidence, and what evidence there is conflicts, on whether people value changes in cancer risks more than changes in other risks. Also, values might be biased by misperceptions of the likelihood of the risks involved. The value attributed to the risk of mortality from cancer is therefore treated the same as for other illnesses (i.e. the standard best estimate).

However, people may be willing to pay more to reduce their risk of dying from cancer because death from cancer may be preceded by a long period of serious illness. This "cancer premium" - relating to the period of ill health prior to death – was thought important to capture. Although evidence on it is minimal, a central assumption for the value of the "cancer premium" is that it is equivalent to 50% of the standard reference values above.

It was noted that values might also change for altruistic reasons relating to context. In other words, individuals' willingness to pay may not be a fair reflection of society's value. However, the conditions for admitting altruistic values in a cost-benefit analysis are restrictive since the requirement is to measuring private WTP for risk reductions. Thus, whilst individuals in a population may place a high value on old people though they themselves do not place this high value on themselves, no adjustment was made for this factor.

In a similar fashion to DG Environment, the UK Government had previously established a working group to derive WTP values for avoiding premature death from air pollution. The recommended values from this are published in The Stationery Office (1999) and are summarized in the following paragraphs.

<u>Baseline VSL</u>
The road transport accident VSL used by the UK Department of Transport was used as a baseline value and is equivalent to approximately €1.2 million.

<u>Context</u>
An adjustment is made to the baseline VSL in order to take account of the involuntariness associated with air pollution impacts relative to road accidents. The adjustment of an increase of between 2 and 3 times is on the basis of the evidence from Jones-Lee and Loomes, (1995) and gives a VSL range of between €2.5 million and €3.5 million.

<u>Age</u>
Adjustments based on the age of the air pollution victim are made based primarily on the Jones-Lee (1989) study and a later unpublished study by the same author – both of which were in the road accident context. The suggested adjustments are:

Table A1: Adjustment of the VSL due to age

Age	Adjusted value (% of context-adjusted VSL)
65	100%
70	80%
75	65%
80	50%
85	35%

An average adjustment of 70% is recommended and, applying the lower-bound context adjusted VSL gives a value of €1.75 million.

<u>Health</u>

Two issues are considered under this heading and we accordingly document them separately:

i) Reduced life expectancy

An argument is put forward that since those affected by air pollution (>65) might have a life expectancy significantly lower than the average for the age group, a reduction should be made to reflect this. An assumption was made that the life expectancy could be one year or one month rather than the average of twelve years. If WTP is then assumed to adjust proportionally, the resulting range of VSLs is therefore between €1.75 million (unadjusted) and €12,000, with a central value of €146,000.

ii) Reduced quality of life

Quality of life indices assumed to be typical to those impacted by air pollution are found to give adjustments of between 0.2 and 0.7, on top of the age group average of 0.76. A value of 1 represents good health. Assuming that WTP falls proportionate to this index, ranges from the life expectancy adjusted values are between €35,400 and €134,500 for the 1-year life expectancy, and €2,900 and €11,050 for the 1 month life expectancy. The upper bound is provided by the age adjusted VSL, unadjusted for life expectancy or quality of life, of €1.75 million.

Final values are therefore:
Lower-bound: €2,900
Mid-bound: €134,500
Upper-bound: €1.75 million

<u>Futurity</u>
Futurity is accounted for by assuming a 1% discount rate per annum – based on an estimate of the pure time preference rate. This implies, for example, that impacts occurring 10 years hence would be valued at 90% of the current value, whilst those occurring 20 years hence would be valued at 82% of the current value.

We summaries the policy values that are currently used in the following Table A2.

Table A2: Summary of values used for policy purposes

Adjustment factor	EC Guideline	UK Govt. Guideline
Baseline VSL	Central: €1.4 million Range: €0.65 - €3.5 million	Central: €1.2 million
Context	50% premium for cancer	Involuntariness – multiply by 2
Age	Multiplier of 0.7 (applies to central value only)	Multiplier of 0.7
Health	No adjustment	Upper estimate: no adjustment. L.E adjustment – multiply by 0.007 or 0.08 Quality of life adjustment – multiply by 0.28 or 0.92
Cultural	No adjustment	No adjustment
Income	No adjustment	No adjustment
Final Unit Values	Central: €1 million Range: €0.65 - €3.5 million	Central: €0.134 million Range: €0.0029 - €1.75 million
Futurity	Discount rate: 4%	Discount rate: 1%

Appendix 2 Country Reports

THE WILLINGNESS TO PAY FOR MORTALITY RISK REDUCTIONS: A SURVEY OF UK RESIDENTS

Anil Markandya, University of Bath, UK
Overall responsibility, scientific input

Alistair Hunt, University of Bath, UK
Technical organisation, survey administration

Anna Alberini, University of Maryland, US
Econometric analysis

Ramon Arigoni Ortiz, University of Bath, UK
Survey administration

<u>Adaptation of the Krupnick survey instrument: Development Protocol</u>
The computerized survey instrument developed by Krupnick et. al. was used for the basis of the pre-testing work. The principle objective of the pre-testing was to identify how best to adapt the survey to the UK context whilst maintaining its comparability with the other European country studies and those undertaken in Canada and the U.S. The need for comparability constrained the scope for changes principally to those in language. Other issues of comprehension were, however, identified.

The UK development work consisted first of a series of ten in-depth interviews with individuals of age 40 and above and an equal gender split. The original US survey instrument was walked-through and issues of comprehension were identified. These interviews were ninety minutes on average. A similar procedure was followed in a series of three focus groups comprising of eight participants in each. A sound recording of each group was made and flip-charts were used during the course of the group sessions as a way of summarizing comments and recapping. The groups were two hours in length and were made up of the same age and gender characteristics as the one-to-one interviews.

The substantive findings of the one-to-one interviews and focus groups – some of which led to changes in the survey instrument – can be summarized in the following points:

- Faced with the questionnaire, respondents tend to think of the "product" as being a medical good. In order to generalise the WTP vehicle in the survey in such a way as to avoid such contextual biases, the testing found that the wording be changed from its use of "product" to "product, or action".

- Understanding of costs of medical action in the UK is complicated by the fact that health care in the post-war has been provided free at point of supply. This is now rapidly changing as more people opt for additional private medical insurance cover. Nevertheless, the testing found that it is important to highlight further the point that cost exists even when an action is free. As a consequence the survey was changed to stress this.

- The issue of there being no stated context was something that a number of respondents were uncomfortable with. The different sized risk changes were discriminated between in most cases, though it was not clear whether respondents were able to process the information to generate WTP in an income-bound context. No changes were made to reflect these as it was judged that such changes would significantly reduce inter-country comparability.

- Some respondents felt that a five-year time period over which the risk change would take place was easier to imagine, and state a WTP value. Also, some respondents felt that they needed evidence that the product would restore full quality of life rather than simply a reduction in the risk of death. For the same reason as the previous bullet point, no changes were made to reflect these comments.

Sampling Frame:
The final survey was conducted in a computer laboratory at the University of Bath where thirty-three groups of ten individuals answered the computerized survey instrument. The total UK sample size was therefore 330. The survey respondents were recruited by a professional recruitment company and were offered a Euro 25 incentive payment for their attendance. They had the remit to recruit on a stratified random basis a sample that closely matched the socio-economic characteristics of the UK population - the area of recruitment for the 328 respondents being a 35 km radius around the city of Bath. The company used a mix of recruitment techniques including random digit dialing, in-street recruiting and snow-balling. Out of 1350 eligible respondents contacted, 355 were "co-operative", and 330 actually attended. Of the 995 that were not co-operative, 560 were not able to travel to the survey centre and 435 did not find the incentive high enough.

Descriptive statistics: Because we cannot claim that the sample is representative of the population of the UK, our first order of business is to examine the individual characteristics of the respondents. Table B.1 displays descriptive statistics of the respondents. The table shows that the composition of the sample is relatively even in terms of gender, that median household income is €42,400, mean household income is almost €44,000, and that our respondents had, on average, about 14 years of schooling, which roughly corresponds to attaining the A levels. Approximately 34 percent of the sample has (private) health insurance, in addition to the national health care. Virtually all respondents identified themselves as white-Caucasian, so no race variables are entered in this table. The experimental design calls for administering the survey questionnaire to persons of age 40 and older, and this requirement is borne out in the data. The minimum age is 40, and the average age is 58 years. The oldest individual in the sample is 77 years old. Roughly 45 percent of the sample is of age 60 or older.

Table B.1. Descriptive Statistics of the Respondents

Variable	Average or Percent of the sample	Standard Deviation	Remarks
MALE	49.39%		
Household income (INCOME) (€)	Mean 43,973 Median 42,400		Midpoints of intervals were used to construct this variable
Income per household member (PCAPPINC) (€)	Mean 18,896 Median 14,000		
Age (years)	58.03	9.26	The oldest individual in the sample is 77 years old
Percentage of respondents in various age groups: Age 40-49 Age 50-59 Age 60-69 Age 70 or older	 20.00% 34.85% 33.33% 11.82%		 Notice that 45.15% of the sample is of age 60 or older
EDUC (years of schooling)	14.10	2.36	17.48% of the respondents has a college degree
ADDLINSUR (has health insurance in addition to national health care)	33.64%		

<u>Objective and Subjective Risks:</u> Table B.2 displays descriptive statistics for three variables. The first is RISK10, the baseline risk of dying over the next years. The average of this variable is 199 (for Caucasians). Respondents were also asked to report their subjectively assessed probability of surviving to age 70. The average of the variable CHANCE70 is only 41.28 percent, which is rather low. The average of the age until they expect to live (AGEDIE) is about 81 years.

Table B.2. Objective and Perceived Risks of the Respondents

Variable	Average or Percent of the sample	Standard Deviation	Remarks
RISK10 (baseline risk of dying over the next 10 years)	198.93 in 1000		This is an objective measure, and is assigned to the respondent based on age and gender
CHANCE70 (chance of surviving until age 70)	41.28	39.15	Subjective--Ranges from 0 to 100
AGEDIE (age until the respondent expects to live)	80.86 years	7.17	Subjective

Health Status: Descriptive statistics about the health status of the respondents are shown in Table B.3. While the rate of chronic respiratory disease (summarized into the indicator LUNGS) is comparable to that of the US, the sample of UK residents appears to have a much lower rate of heart problems (only 8 percent, compared to 10% and 21% for Canada and the US). The percentage of respondents who states that they are in excellent or very good health relative to others the same age is just slightly higher than in Canada and the US (53% and 57%, respectively).

Table B.3. Health Status of the Respondents

Variable	Average or Percent of the sample	Remarks
CARDIO (any of coronary, angina, heart attack, or other heart disease)	8.18%	
LUNGS (any of emphysema, chronic bronchitis or asthma)	15.45%	
PRESSURE (high blood pressure)	28.48%	
CANC (has been diagnosed with cancer)	6.36%	
CHRONIC (any of CARDIO, LUNGS, PRESSURE, or has suffered a stroke)	43.33%	Note that the construction of this variable does not include cancer
ER_HOSPITAL (has visited emergency room or has been hospitalized in the last 5 years for respiratory or heart problems)	6.67%	
GOODHEAL (respondent judges his/her health to be very good or excellent relative to others the same age)	60.79%	

<u>Probability Comprehension:</u> Because the survey instrument is about probabilities and changes in probabilities, it is important to examine respondent facility with probabilities. Table B.4 shows that 15 percent of the UK sample failed the so-called probability test, which asks which person, A or B, has the higher risk of death. Most respondents, however, corrected themselves when prompted for a confirmation of their answer. Less than 1 percent of the sample (3 subjects) insisted on the wrong answer. In addition to this probability quiz, the questionnaire also contains a probability choice question: Given two individuals, A and B, facing different risks of death, which would the interviewee rather be? Fourteen percent of our subjects chose the person with the higher risk of death, but once again almost all of them changed their minds when prompted to confirm. Finally, about 27 percent of the sample feels that they understand the concept of chance poorly. This figure is higher than in any of the previous studies.

Table B.4. Probability comprehension

Description	Percent of the sample
Answers the probability test wrong	15.33
Confirms wrong choice to the probability test	0.91
Shows preference for the person with the higher risk	14.29
States he/she is indifferent between the lower and higher risk person	7.00
Confirms preference for higher risk person	1.52
Understands probability poorly (FLAG6=1)	26.97

<u>Comprehension of the Survey Instrument</u>: Table B.5 reports descriptive statistics for indicators based on the respondents' answers to the debriefing questions at the end of the survey. Briefly, the UK is similar to the Canada and US samples in terms of their reactions to many aspects of the questionnaire. It should be noted, however, that (i) more of the UK respondents reported a poor understanding of the concept of probability (FLAG6), (ii) fewer of the UK respondents had considered other benefits of the product, (iii) the UK respondents are less likely to say that they did not even considered whether they could afford the product described in the survey (FLAG15), and that (iv) failure to understand the payment scheme (FLAG16) is less likely to occur with the UK respondents.

Table B.5. Debriefs in the UK mortality risk study

FLAG	Description	Percent of the sample with FLAG equal to 1
FLAG1	Wrong answer to the prob. test and chooses person with higher risk	2.45
FLAG2	Flag1=1 but respondent does not confirm preference for higher risk	2.12
FLAG3	Answered first probability test wrong	15.15
FLAG4	Answered second probability test question wrong	0.91
FLAG5	Confirmed preference for higher risk	1.52
FLAG6	Understands chance poorly (selects 1-5 on a scale from 1 to 7, where 1 is worst understanding and 7 is best understanding)	26.97
FLAG7	Did not believe risks	20.91
FLAG8	Has doubts about the effectiveness of the product	34.55
FLAG9	Doubts about the effectiveness of the product influenced WTP	20.30
FLAG10	Thought about side effects	16.67
FLAG11	Considered other benefits of the product or did not know	32.12
FLAG13	Considered (as he should have) the chance of living to and health at age 70	94.85
FLAG14	Did not understand that payment would begin this year	13.64
FLAG15	Did not consider whether he could afford payments	20.91
FLAG16	Did not understand payment scheme	3.64

Responses to the payment questions: We use a dichotomous-choice approach with two follow-up questions to elicit information about the respondent's WTP for specified risk reduction. The second follow-up question is asked only of those individuals who declined to pay both the initial and follow-up bid amounts (see Table B.6), and attempts to find out if the respondent holds a positive, but low, WTP, or if WTP is zero.

Table B.6. Initial and follow-up bids in the UK study (€)

Initial bid	Bid if response to first payment question is yes	Bid if response to the first payment question is no
70	160	30
160	520	70
520	760	160
760	1040	520

Table B.7 displays the percentages of the samples who answered "yes" to the different bid values for the initial payment question. Clearly, for the 5 in 1000 risk reduction, the percentage of "yes" responses falls with the bid amount, implying that the individual responses are consistent with economic theory. It is troublesome that this desirable pattern is not observed in the responses to the payment questions for 1 in 1000 and future risk reductions. (See Figures B.1 and B.2 for a graphic presentation of these results.) It is comforting, however, that the percentages of "yes" responses, however, are less for the smaller and future risk reduction than for the 5 in 1000 risk reduction, which suggests that the estimates of WTP are likely to pass the so-called scope test.

Table B.7. Distribution of the "Yes" responses to the initial payment question

Initial bid € (British pounds)	Commodity being valued		
	5 in 1000 risk reduction over 10 years starting now (1st commodity) n=330	1 in 1000 risk reduction over 10 years starting now (2nd commodity) n=330	5 in 1000 risk reduction over 10 years starting at age 70 (3rd commodity) n=187*
70 (45)	71.11	36.67	36.00
160 (100)	70.73	42.68	45.45
520 (325)	48.75	17.50	19.61
760 (475)	41.03	24.36	19.05

* This question was asked only of respondents younger than 60 years of age.

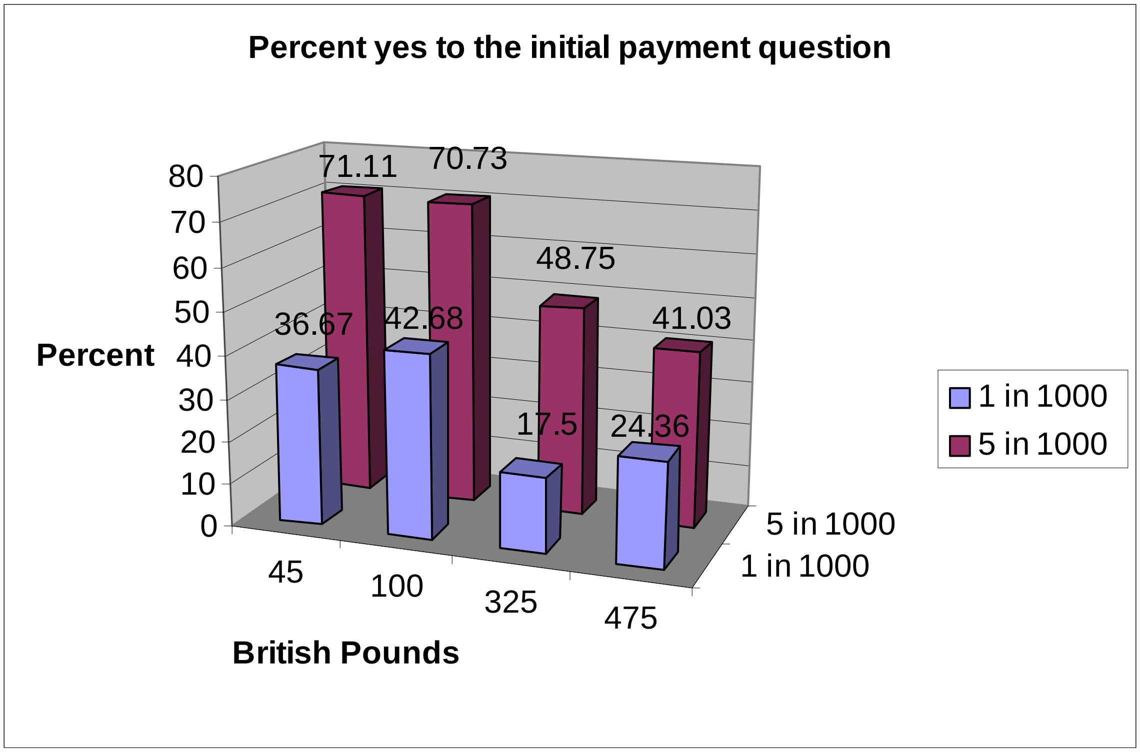

Figure B.1. Responses to the initial bid question for the immediate risk reductions.

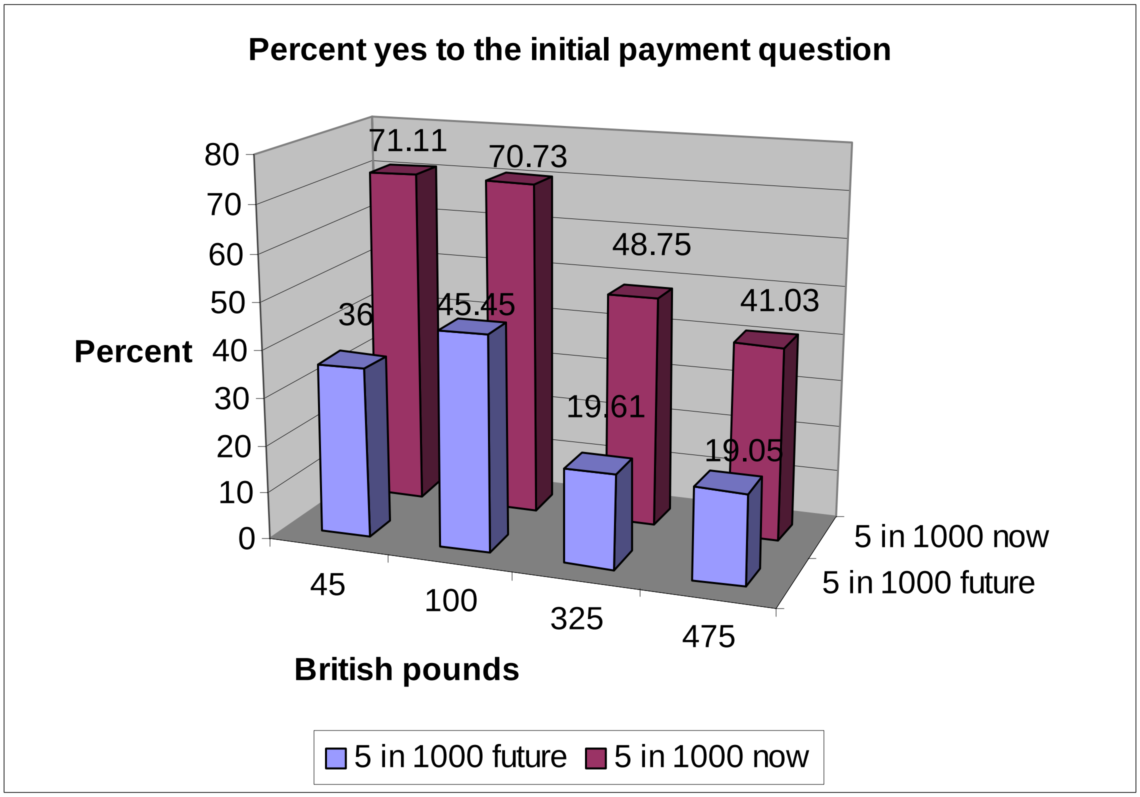

Figure B.2. Responses to the initial payment question for the 5 in 1000 risk reduction, effective immediately and effective at age 70.

In Table B.8 we examine the proportions of respondents with WTP equal to zero. These are 15.76 percent for the 5 in 1000 risk reduction, 42.12% for the 1 in 1000 risk reduction, and 41.71% for the future risk reduction.

Table B.8. Percent respondents who report have WTP equal to 0

Commodity being valued		
5 in 1000 risk reduction over 10 years starting now (1st commodity) n=330	1 in 1000 risk reduction over 10 years starting now (2nd commodity) n=330	5 in 1000 risk reduction over 10 years starting at age 70 (3rd commodity) n=187*
15.76	42.12	41.71

Estimates of mean and median WTP are reported in Table B.9. These estimates are based on a fully parametric model that assumes that WTP follows the Weibull distribution and forms intervals around the respondent's WTP amount using the responses to the initial and first follow-up questions (ignoring the second follow-up for those respondents with no-no responses). We present results for (i) the full sample, (ii) a "cleaned" sample that excludes individual who failed the probability test twice (FLAG1=1), and (iii) a sample with individuals that feel strongly about their WTP for the 1 in 1000 risk reduction.

Table B.9. Estimates of WTP (standard errors around mean or median WTP in parentheses) (€)

Risk reduction	All sample (n=330)	Flag1=1 deleted (n=322)	Only respondents who state that they have certainty level higher than 6 in their response to the WTP questions for the 1 in 1000 risk reduction* (n=153)
5 in 1000 risk reduction			
mean WTP	722 (91.3)	736.3 (100.2)	787.6 (165.9)
median WTP	386.3 (36.3)	387.6 (37.9)	302.3 (47.9)
1 in 1000 risk reduction			
mean WTP	334.4 (54.4)	330.8 (52.9)	277 (1170.2)
median WTP	90.4 (13.6)	88.2 (13.9)	31.2 (12.1)
Future risk reduction	(n=187)	(n=182)	(n=86)
mean WTP	313.6 (53.8)	302.3 (54.3)	346.9 (256.5)
median WTP	113.9 (19.2)	111.1 (19.2)	67.9 (23.8)

* persons were selected who stated they had a certainty level of 6 or 7 when answering the payment questions for the 1 in 1000 risk reductions because doing so afforded the most usable observations. The number of respondents who indicated a high level of certainty (6 or 7 on a scale from 1 to 7, with 1 indicating the least certainty and 7 the highest certainty) when answering the payment questions was 99 for the 5 in 1000 risk reduction (30% of the respondents), 153 for the 1 in 1000 risk reduction (46.67%), and 84 for the future risk reduction (44.92%).

As shown in Table B.10, WTP passes the (internal) scope test, but the evidence about proportionality to the risk reduction is mixed. Only median WTP passes the proportionality test, which states that WTP for the 5 in 1000 risk reduction should be 5 times that for the 1 in 1000 risk reduction, and when attention is restricted to those respondents who feel very strongly about their answers to the payment question, there is some possible evidence of over-proportionality. We believe that the latter results is probably due to the fact that respondents tend to feel strongly about "no" responses, and are more lukewarm about their "yes" responses to the payment questions, especially with the 1 in 1000 risk reduction. While this was observed in the Canada and US studies as well, we feel that caution should be used in its interpretation.

Table B.10. Internal scope and proportionality tests

Question	All sample (n=330)	Flag1=1 deleted (n=322)	Respondents who are certain of their answers (see def. In Table B.9) (n=153)
INTERNAL SCOPE TEST: Is WTP for the 5 in 1000 risk reduction greater than WTP for the 1 in 1000 risk reduction?	Mean WTP: YES (Wald test is 13.28, P value for chi square with 1 dof < 0.001)—scope test passed median WTP: YES (Wald test is 58.21, P value for chi square with 1 dof < 0.0001—scope test passed	Mean WTP: YES (Wald test is 13.00, P value for chi square with 1 dof < 0.001)—scope test passed median WTP: YES (Wald test is 55.02, P value for chi square with 1 dof < 0.0001—scope test passed	Mean WTP: This test was not performed due to the unreliable estimate of mean WTP for the 1 in 1000 risk reduction in this group Median WTP: YES (Wald test is 30.13, P value for chi square with 1 dof < 0.0001—scope test passed
INTERNAL PROPORTIONALITY TEST: Is WTP for the 5 in 1000 risk reduction <u>5 times</u> WTP for the 1 in 1000 risk reduction?	Mean WTP: NO (Wald test is 10.96, P value for chi square with 1 dof < 0.001—fails proportionality test) RATIO=2.16 Median WTP: YES (Wald test is 0.73, P value for chi square with 1 dof is 0.39—proportionality test passed) RATIO=4.27	Mean WTP: NO (Wald test is 10.54, P value for chi square with 1 dof < 0.001—fails proportionality test) RATIO=2.22 Median WTP: YES (Wald test is 0.45, P value for chi square with 1 dof is 0.50—proportionality test passed) RATIO=4.39	Mean WTP: This test was not performed due to the unreliable estimate of mean WTP for the 1 in 1000 risk reduction in this group Median WTP: BARELY (Wald test is 3.58, P value for chi square with 1 dof is 0.058)(Notice that the ratio of the two median WTP amounts is 9.69, which suggests overproportionality) RATIO=9.69

Table B.11 presents the VSL figures (based on the cleaned sample)

Table B.11. Annual value of a statistical life based on the figures of Table B.9 for the sample without respondents with FLAG1=1 (€)

	From WTP for the 5 in 1000 risk reduction from age 70	From WTP for the 1 in 1000 risk reduction
Using mean WTP	967,360 (173760)	3,308,160 (529,280)
Using median WTP	355,520 (61440)	881,920 (138,560)

* These figures are computed by taking the annual WTP figures and dividing by X/10000, where X is the risk reduction, which is assumed to be evenly spread over 10 years. This approach eliminates the need for choosing a discount rate.

** Standard errors in parentheses.

<u>WTP Regressions</u>: Regressions that check the internal validity of the responses and examine the effect of various factors on WTP have been made for the 5 in 1000 risk reduction and the WTP for the future risk reduction. We assume that WTP, which is not observed directly, follows the equation:

(1) $$\log WTP_i = x_i \beta + \varepsilon_i ,$$

where **x** is a vector of individual characteristics and risk variables, β is a vector of unknown parameters, and the error term follows the type I extreme value distribution. Willingness to pay, therefore, follows an accelerated-life Weibull model.

The vector **x** contains age and health status, socio-demographic variables such as gender, education, income and health insurance, and the health status of relations, which may account for familiarity with illness and may affect WTP. In certain specifications we also include baseline risk, or the respondent's subjective remaining life. When we examine WTP for future risk reductions, the vector of regressors also include the respondent's subjective probability of surviving until age 70 (when the risk reduction would be incurred) and his expected health status at that age.

Focussing on the effect of age on WTP for the 5 in 1000 risk reduction, different functional forms were tried, but we detected no meaningful association between respondent age and WTP. We then check if WTP depends on remaining life. The coefficient of remaining life is, in fact, positive but insignificant. Possible associations between WTP and baseline risk are checked but neither absolute nor proportional baseline risk are significantly associated with WTP, and the coefficient on the former of the wrong sign (negative). In regressions not reported, we checked if controlling for other variables changed these results, but did not find that to be so.

In a specification where age is controlled for using age dummies, and individual characteristics of the respondent are added, we find that higher education levels tend to be associated with lower WTP amounts (an effect seen in the Canada and US studies as well, although not statistically significant), and that income per household member is positively and significantly associated with WTP. In general, the health status did not matter, although the coefficient of the CHRONIC dummy was positive and significant at the 6% level, which further controls for a chronic illness in the family, cancer among relations, and additional health insurance. The coefficient of CHRONIC is 0.18, implying that suffering from any of the CARDIO, LUNGS or PRESSURE illness, or having had a stroke, tends to raise WTP by about 20 percent. It is surprising that the presence of illness in the family tends to be negatively associated with WTP.

Results tend to be robust to the inclusion of indicator (the FLAG variables) based on the debriefing questions. WTP does not appreciably change with the respondent's refusal to believe the risk figures (FLAG7) (although the coefficient on this variable has a negative sign, as one would expect), but is much lower for individuals who doubted the effectiveness of the product. Thoughts about the product's side effects do not influence WTP, but WTP is much lower for individuals who did not even think whether they could afford the product. This should be interpreted as suggesting that individuals who had already ruled out purchasing the product did not even bother to think whether they could afford the payments in the first place. Finally, those persons who misunderstood the timing of the payments tend to have a lower WTP, although this is not a fully significant effect.

Regarding WTP for the future risk reduction, we find that it tends to increase with the (log) chance of surviving to age 70 and to decrease if the individual thinks his or her health will be worse in the future. Other variables do not matter, with the only exception of the presence of chronic illness among relations.

THE WILLINGNESS TO PAY FOR MORTALITY RISK REDUCTIONS A SURVEY OF FRENCH RESIDENTS

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Serge Masson, BETA, Université de Strasbourg,
Translation and adaptation of questionnaire, technical organisation, administration of the questionnaires

Ari Rabl, Ecole des Mines de Paris
Scientific input, calculation of life expectancy gain, final report

Laure Santoni et Marie-Anne Salomon, EdF,
Variants public good and life expectancy

Summary

This report describes the application of the questionnaire of Krupnick et al in France, as part of WP 2 of the NewExt Project. The original questionnaire was ad-ministered to 300 individuals, but by contrast to the application in the UK and Italy, an open question was added after each set of bids; at the end of the questionnaire the WTPs were recalled to give the respondents the opportunity to correct their values. In addition several variants were tested on samples of about 50 each, in particular variants phrased in terms of life expectancy gain. All the interviews (self-administered with a computer) were followed by written in-depth debriefing, and for the two last variants by face-to-face debriefing and discussions in groups of three or four, in order to better understand the perception of the questionnaire and the reasons for the responses. The results are used to provide estimates for the value of statistical life (VSL) and for the value of a life year (VOLY): they range from 0.4 to 4.4 M€ for VSL and from 0.020 to 0.220 M€ for VOLY. However, the most important results are not the numbers but the lessons learned by debriefing and by the variants of the questionnaire. The wide range of results for VSL and VOLY is a reflection of the enormous difficulties that the respondents have in understanding risk reductions and replying to the WTP question. Thanks to the open question it was possible to measure the bias due to the starting bid: it is very large, on the order of 50% for the bids that were used. Thus the recommendation of the NOAA Panel on contingent valuation, that only the closed question should be used, is not appropriate for mortality.

The Sample

Interviewees were recruited by a private marketing firm, using the French quota system to be representative, in age, sex and income, of the population of Strasbourg. They were paid 20 € to come to the Experimental Economics Laboratory of the University of Strasbourg.

- Total size of the sample 299
- 151 answering the sequence S1 (first 5/1000 then 1/1000),
- 148 answering the sequence S2 (first 1/1000 then 5/1000).

Bids offered, in French francs, 500 FF, 1000 FF, 3500 FF, 5000 FF; if no to 500 FF second offer is 200 FF, if yes to 5000 FF second offer is 7000 FF (1 FF = 0.15 €).
We offered the same starting value for the different risk reductions.

The characteristics of the sample are summarized in Tables C.1 and C.2.

Table C.1. Structure of the sample by sex, age and starting bid.

	S2 (1/1000 then 5/1000)					**S1 (5/1000 then 1/1000)**			
TotalMen	142	500 FF	1000 FF	3500 FF	5000 FF	500 FF	1000 FF	3500 FF	5000 FF
No. Aged 40-50	53	6	6	6	8	6	7	6	8
No. Aged 51-60	43	8	5	4	4	6	5	6	5
No. Aged 61-75	46	5	5	6	4	8	8	5	5
TotalWomen	157	500 FF	1000 FF	3500 FF	5000 FF	500 FF	1000 FF	3500 FF	5000 FF
No. aged40-50	63	9	9	8	7	7	7	8	8
No. aged51-60	40	6	5	5	5	6	5	4	4
No. aged61-75	54	7	6	7	7	6	7	7	7
Total		41	36	36	35	39	39	36	37
Total	299	**Total of S2**		148		**Total of S1**		151	

Table C.2. Characteristics of the respondents.

	S2 (N = 148)			S1 (N = 151)			Total (N = 299)		
Characteristics	Mean	Median	Stand.dev.	Mean	Med.	Stand.dev.	Mean	Median	Stand.dev.
Age in years	55.0	53.5	10	55.7	55.1	10	55.4	55	10
Female	54.7%			50.3%			52.5%		
Education	3.8	3	2.2	3.6	3	2.1	3.7	3	2.1
Household income	3.6	3	1.6	3.7	4	1.4	3.6	4	1.5
Mental health score	47.0	50.3	13	48.1	51.5	11.8	47.6	51.1	12.5
Physical Functioning score	45.8	47.5	6.8	46.2	47.2	6.7	46.0	47.3	6.7
Baseline risk over 10 years	103.7	73	85.5	114.2	77	93.3	109	77	89.5
Heart disease	14.2%			18.5%			16.4%		
High blood pressure	25 %			17.2%			21.1%		
Cancer	6.8%			6.0%			6.4%		
Asthma	10.1%			10.6%			10.4%		
Bronchitis, emphysema, or chronic cough	16.9%			11.9%			14.4%		
Self-assessed life expectancy, in years	31.5	31	11.5	31.6	31	10.5	31.6	31	11.0

Income: 8 categories of household income

Category	Income, FF/month	Number of respondents
1	less than 6000	19
2	6000 to 10 000 francs	64
3	10 000 to 15 000 francs	61
4	15 000 to 20 000 francs	61
5	20 000 to 30 000 francs	63
6	30 000 to 40 000 francs	21
7	40 000 to 50 000 francs	7
8	over 50 000 francs	3

Education: 9 categories

Category	Description	Number of respondents
1	Primary school (4 years)	40
2	Secondary school (11 years)	67
3	Secondary school (11 years) with diploma for university	43
4	Technical or professional school (11 years)	53
5	2 years university	36
6	4 years university	23
7	5 years university	15
8	5 years technical university	9
9	Doctorate	13

No question is asked about race (controversial in France)

Number of children	Number of respondents
0	48
1	62
2	103
3	53
More than 3	33

An important point: 269 persons (90%) have private supplementary health insurance to supplement the social security system, so 80 to 100% of their medical expenses are reimbursed and they have to pay only 0 to 20% themselves; this means that they have little or no awareness of the costs.

Health status
51% of the sample think that their health is comparable to the rest of their age group, 38 % that it is better than the rest of their group, and 11% than it is worse; this asymmetry between 38 % and 11% is a sign of optimism. But 10 years from now 52% think that their situation will worsen, 48% think that their situation will be comparable.

Results for Risk Comprehension, Scenario Acceptance and Payment

Risk comprehension
As shown by Table C.3, the respondents have trouble understanding probabilities (23% failed the first probability test, 22% chose the wrong person in probability test and the wrong person in probability choice) but learn fast to correct their answer. 18 % acknowledged a poor comprehension of probabilities and it is certainly an underestimation of reality.

Table C.3. Risk comprehension

Sequence	1/1000 then 5/1000 (N =148)		5/1000 then 1/1000 (N =151)		Total (N = 299)	
Percent of respondents who….	%	Number	%	Number	%	Number
chose wrong person in first probability test	24%	35	22%	33	23%	68
chose wrong person in second probability test	2%	3	6%	9	4%	12
chose wrong person in probability choice	14%	21	7%	10	10%	31
have no preference	22%	32	23%	35	22%	67
chose wrong person in probability test and wrong person in the probability choice	4%	6	0%	0	2%	6
chose wrong person in probability test and have no preference between the two individuals	9%	14	9%	13	9%	27
confirm wrong person in probability choice	2%	3	1%	1	1%	4
chose wrong person in second probability test and wrong person in the probability choice	0%	0	0%	0	0%	0
indicate 3 or less in self-assessed understanding (on a scale of 1-7)	14%	20	22%	33	18%	53

<u>Scenario acceptance</u>

Verbal protocols and debriefing have shown that French people tend to doubt the efficacy of a product that is not recognized and reimbursed by the Social Security system. This is reflected in the Table C.4.

Table C.4. Scenario acceptance

Sequence	1/1000 then 5/1000 (N =148)		5/1000 then 1/1000 (N =151)		Total (N = 299)	
Percentage of respondents who….	%	Number	%	Number	%	Number
do not believe the stated risks apply to them	11%	17	10%	15	11%	32
have doubts about the product's effectiveness (or don't know)	38% (16%)	56 (23)	38% (19%)	55 (28)	38% (17%)	111 (51)
have doubts about the product's effectiveness and said doubts affected WTP	18%	26	18%	17	18%	53
think product might have side-effects	46%	68	43%	65	44%	113
think about other benefits of the product	23%	34	28%	43	26%	77
say other benefits influenced WTP	35%	12	40%	17	38%	29
think that the product decreases only the risk of dying	30%	91	58%	87	60%	178

The payment

Table C.5 shows what the respondents say to the questions about their payment. 24% of the sample do not take into account their budget constraint (18% + 6% I don't know, 72 persons). We must discriminate between those who did not take in account their budget because they refuse to pay (39 persons who probably reject the scenario), and those who gave a positive value (33 persons with a mean WTP of €551).

Table C.5. Questions referring to the payment

Sequence	1/1000 then 5/1000 (N =148)		5/1000 then 1/1000 (N =151)		Total (N = 299)	
Percentage of respondents who ….	%	Number	%	Number	%	Number
understand correctly that the amount stated would be paid during 10 years	96%	142	96%	145	96%	287
take into account their budget	72%	107	79%	120	76%	227
do not take into account their budget	22%	33	15%	22	18%	55
do not know	6%	8	6%	9	6%	17
think about other benefits of the product	23%	34	28%	43	26%	77
say other benefits influenced WTP	35%	12	40%	17	38%	29
do not consider whether they could afford the payment or don't know	28%		20%	31	24%	72
do not understand the payment scheme or don't know	4%	6	3%	6	4%	12

Response to the payment question

The answer to a double bid is difficult (as shown by verbal protocol and debriefing). For many respondents it was the first time they used a computer. Even if the use was made as simple as possible we learned (too late) that the procedure to correct a false number was too difficult for many of them. One of the problems was that some respondents could not read correctly what they typed, for example could not distinguish between 10000 and 100000. We offered, at the end of the questionnaire, the opportunity to correct the amounts stated. Many used this opportunity. So we could detect some of these typing errors.

Another very common typing error occurred with the answer to the bids. For example a person was offered 3500 FF, wanted to say no but typed yes, was offered a higher bid of 5000 FF, typed correctly no, and then stated a WTP of say 1000 FF. By looking carefully at the answers we could detect 22% of these two kinds of typing errors.

Concerning the error in typing the values, we offered the respondents to write the figures on a paper showing the number of the computer (for the last variants) so we could compare the typed and the written values.

Consistency of the answers

The following comments are based on the answers to the open question posed after the responses to the bids. At the end of the questionnaire, respondents could see in a recall table the three (or two) values stated, and correct them if wanted. For many of them, with poor memory, it was the opportunity to see the inconsistency of their answers and correct them.

Sequence S1 (first 5/1000 then 1/1000) : 151 respondents
We expect that WTP decreases with the decrease of probabilities. The third value stated by respondents 40 to 60 years old is supposed to express a kind of assurance they buy today to increase their life expectancy.
We observe that

2 64 give the same value (including 21 with zero WTP), when offered 22 persons will correct their WTP, becoming more consistent.
3 75 are consistent, 9 will correct their WTP and give the same value.
4 12 are inconsistent (higher value for 1/1000), 5 will correct their WTP

36 persons correct their WTP, generally by lowering it, and/or becoming more consistent.
23 (of 98) persons give a null WTP for the last decrease in probability.

Sequence S2 (first 1/1000 then 5/1000) : 148 respondents
We *expect* that WTP increases with the increase of probabilities.
We *observe* that :

5 93 give the same value (including 34 with zero WTP), 36 persons correct their WTP.
6 43 are consistent, 16 correct their WTP, 12 giving now the same value.
7 12 are inconsistent, 11 correct their WTP

63 persons correct their WTP, trying to become more consistent, or changing (generally lowering) the different values stated. 33 persons (of 101) give a null WTP for the third decrease in probability.

Conclusion

The second wave shows a higher number of zeroes, the mental exercise seems to be more difficult and we observe less consistency in the answers with a higher number of equal values. Moreover the stated values appear more unstable. The starting probability was perceived as too low. Respondents who stated a positive WTP could not increase their WTP further. So they had to correct more often the first stated value.

Answers to the proposed bids

Table C.6. Answers to the proposed bids

	S1		5/1000 70 to 80	S2		5/1000 70 to 80
	1/1000	5/1000	5/1000	5/1000	1/1000	5/1000
NO NO followed by a positive amount	24 (16%)	18 (12%)	8 (8%)	14 (9%)	14 (9%)	12 (12%)
NO NO followed by zero	43 (29%)	34 (23%)	33 (33%)	21 (14%)	51 (34%)	23 (23%)
YES (NO)/ NO (YES)	33 (22%)	42 (28%)	31 (31%)	28 (19%)	37 (25%)	30 (31%)
YES YES	48 (32%)	54 (36%)	29 (29%)	88 (58%)	47 (31%)	33 (34%)
Total	148	148	101	151	151	98

Table C.7. Percentage of yes to the initial bid (in parentheses the numbers after correction of typing errors).

	First risk proposed		Second risk proposed		5/1000 from 70 to 80	
Initial bid amount	1/1000 (S2)	5/1000 (S1)	1/1000 (S1)	5/1000 (S2)	5/1000 (S1)	5/1000 (S2)
500 FF	56% (59 %)	82% (82%)	54% (56%)	63% (63%)	68% (68%)	48% (48%)
1000 FF	50% (47%)	82% (81%)	67% (62%)	56% (53%)	67% (67%)	56% (56%)
3500 FF	33% (29%)	67% (50%)	39% (36%)	39% (31%)	42% (42%)	26% (26%)
5000 FF	31% (29%)	54% (44%)	19% (19%)	51% (44%)	28% (28%)	38% (38%)

It is interesting to note that we observe almost the same percentage of yes for 500 as for 1000 FF and for 3500 as for 5000 FF for the first proposed risk. We suspect an anchoring bias, and that people consider these bids as equal, which means a large personal range of uncertainty about the "real" value. The English data show the same phenomenon. Even if we take in account the typing errors, we stay in the same range of values

Analysis

Some results of the econometric analysis are listed in Tables C.8 and C.9.

Table C.8. Mean WTP estimated by logit or spike of double bid, or by non-parametric estimation, compared with mean WTP of open question.

	Mean of open question (all WTP)	Mean of open quest. [d] (WTP <20000FF)	Logit of bids	Spike of bids	Non-param. of bids	Mean of open question (all WTP)	Mean of open quest. [e] (WTP <20000FF)	Logit of bids	Spike of bids	Non-param. of bids
Risk reduction	1/1000	1/1000	1/1000	1/1000	1/1000	5/1000	5/1000	5/1000	5/1000	5/1000
Sequences										
S1 [a]	419 €		347 €	434 €		649 €		592 €	906 €	
(standard dev.)	*(822 €)*					*(791 €)*				
S2 [b]	404 €		285 €	446 €		476 €		375 €	541 €	
(standard dev.)	*(664 €)*					*(738 €)*				
S1+S2 merged [c]	412 €	322 €	294 €	440 €	208 €	563 €	486 €	482 €	712 €	287 €
(standard dev.)	*(746 €)*	*(456 €)*	*(24 €)*			*(769 €)*	*(540 €)*	*(62 €)*		

[a] S1 : 5/1000 followed by 1/1000 (151 respondents)
[b] S2 : 1/1000 followed by 5/1000 (148 respondents)
[c] S1 + S2 merged : 299 respondents
[d] after removing 7 outliers
[e] after removing 6 outliers

Note that the ratio (WTP for 5/1000)/(WTP for /1000) is 1.6 for S1, and 1.2 for S2. It is easier to lower an initial value, than to increase it. At which point are people able to distinguish between 1/1000 and 5/1000?
Among the many regressions of the open question that we tried, the only variables found to be consistently significant are the initial risk reduction, the starting bid, the income and the self-assessed certainty of the answer. Table C.9 shows typical results, separately for all WTP and for positive WTP (in either case after eliminating outliers of 20000 FF or higher). This separation is appropriate because the respective populations are quite different. It is interesting to note that WTP seems to increase with age but at a decreasing rate; however the coefficients are not significant.

Table C.9. Typical results of linear regression of the open question WTP (in FF) for the 5/1000 risk reduction (292 observations)

	all WTP<20000 R2 = 0.11		**only 0<**WTP<20000 R2 = 0.14	
	Coefficients	t	Coefficients	t
Intercept	-4536	-0.7	-3598	-0.5
Risk reduction (dummy S1 = 1)	1060	2.7	1022	2.3
Starting bid	0.27	2.5	0.40	3.4
Income	492	3.7	478	3.4
Certainty	180	1.9	275	2.6
Age	120	0.5	77	0.3
Age^2	-0.90	-0.4	-0.44	-0.2

This regression reveals a serious bias problem: increasing the starting bid by X increases the positive WTPs by 0.4 X on average. For example, their average goes up by 1800 FF when the starting bid is changed from 500 FF to 5000 FF; this is a very large effect, considering that the average WTP for the 5/1000 risk reduction for the entire set of 299 respondents is less than 4000 FF. Being totally unfamiliar with the valuation of mortality, people are strongly influenced by the starting bid. This shows the limits of the recommendation of the NOAA Panel on contingent valuation [Arrow at al 1993] that the closed question should be used, a recommendation followed by almost all contingent valuation studies since then, without testing its relevance.

The starting risk reduction also has a large impact: on average the positive WTPs are 1022 FF larger when the starting risk reduction is 5/1000 rather than 1/1000. This highlights the difficulty that the respondents have with the concept of risk reduction.

What the Figures Do Not Tell

Verbal protocol, in-depth debriefing, and implementation of variants on small samples (50 persons each) has shown the fragility (volatility) of the above results.

1) The commodity valued

There are several serious problems with the current version of the questionnaire:
To construct their WTP the respondents try to put the proposed medical product or treatment and the risk reduction in perspective (as the base line risk differs by sex and age). Someone who thinks of cancer or AIDS will not state the same value as someone who thinks of vitamins or other dietary supplements: the commodity valued is not homogeneous over the sample.

Under the French social security system most individuals have little or no awareness of the cost of a medication or treatment. Note that 90 % of the sample have private supplementary insurance to supplement the Social Security system, and pay only 0 to 20% of the bill themselves.

The reference value used to construct an answer is more probably a medication or treatment that is not reimbursed by Social Security. However, a product not reimbursed by Social Security is considered less effective than one that is reimbursed. Also, we do not know how the respondents put their risk reduction in perspective and what they think of the corresponding gain in life expectancy.

Furthermore for many respondents in good health, offering them to take a medication during 10 years was the first reason to reject the scenario and refuse to pay anything. For them the "price to be paid" was to high.

2) The elicitation question

A proposed bid can serve as reference to help the respondents determine their answer. But here the debriefing showed that if the bid offered was not consistent (too high) with the product or treatment imagined by the respondents, they were disturbed, thinking that they had misunderstood the scenario; they tended to jump to a different product or treatment and were more inclined to say "yes" than "no".

3) Temporality

People understood that they had to take the medication (every day implicitly) during 10 years. But they did not understand that the risk reduction was for the entire 10 year period, i.e. 1/10000 or 5/10000 per year. This is another weakness of the questionnaire which can be avoided by stating directly the individual gain in life expectancy.

Thus the debriefing shows that the original questionnaire suffers from several problems with anchoring bias:

1 one related to the medication or treatment : or a person accept this payment vehicle and think to a medication, or he/she reject the scenario and refuse to pay anything for the medication, but not necessarily for a reduction in their risk of death. The reasons for rejection can be : a) the person is opposed to the idea of taking a medication during 10 years, b) or has doubt about the efficiency of a medication not reimbursed by the Social Security system, c) some refuse to pay for something that is not sufficiently well specified.

2 one related to the risk reduction : some people tend to think very simply in terms of death or live, not in risk reduction of death, and we can suspect that the values stated are higher.

3 one related to the bid offered : as people do not have a clear opinion upon the "price" of the medication which can "save" their life, they tend to anchor their values to the offered bid.

The Variants

To test the sensitivity of the results to the commodity proposed, to the elicitation question and to the risk reduction, we carried out several variants on samples of about 50 persons each.

Variant 1: Public Good
For this variant, administered by EdF, one sentence was changed in the questionnaire: the good to be valued was described as a public health policy instead of a product (or treatment), and only one first bid was offered (1000 FF) due to the small size of the sample (52 persons), followed by an open question. The payment vehicle is an increase of the social security contribution which is deducted from the salary and collected and administered by a public agency. The anchoring bias diminishes but we do not know how much. The results are shown in Table C.10.

Table C.10. Mean WTP for the variant Public Health.

Risk reduction Sequence	1/1000	5/1000	5/1000 from70 to 80
S1	140 €	327 €	137 €
S2	220 €	239 €	229 €
S1+S2 merged	180 €	283 €	183 €

S1 : 5/1000 followed by 1/1000
S2 : 1/1000 followed by 5/1000

Ratio (WTP for 5/1000)/(WTP for 1/1000) for S1 is 2.3
Ratio (WTP for 5/1000)/(WTP for 1/1000) for S2 is 1.1

Mean WTPs are lower than those for the original questionnaire, by a factor of about 2, but the credibility of the scenario is enhanced: only 17% of the respondents doubt the effectiveness of the public policy (compared to 37 % for the product), and 90% take into account their budget constraint (compared to 76 % for the product). The number of zero WTPs is lower: 12 % (8% in S2 and 15% in S1). But 60% thought of other benefits (mostly for society in general), compared to 26 % for the product. The debriefing showed that people adopt more easily a free rider strategic behavior ("I prefer that the government increases taxes on alcohol and tobacco"), which is comprehensible because in France nobody can be excluded from the benefits of the health system.

Variant 2: Open Question without Bids
In this version the bids are removed and only the open question is asked. This version was applied to a sample of 50 persons answering the S2 sequence (1/1000 followed by 5/1000), and followed by in-depth face-to-face debriefing. The results are shown in Table C.11.

Table C.11. Mean WTP with open question. Sequence S2.

Risk reduction	WTP	% of 0
1/1000	196 €	33%
5/1000	256 €	24%
5/1000 from 70 to 80	209 €	50%

Was this mean value parametric or simply an average of the values? If parametric, where is the econometric result?
Now only 3 respondents reject the scenario ("I don't want to take a medication during 10 years", "a medication not reimbursed cannot be efficient", "a treatment cannot be blind, for which illness ? that is even not mention in your questionnaire") . We have a higher number of zeros : 33% for 1/1000 and 24% for 5/1000. There is no anchoring bias, but the S2 sequence yields lower values, as we saw in the basic questionnaire.

Variant 3: Life expectancy
This variant was tested in three versions:

i. Only the gain in life expectancy (LE) was stated; this version was applied to 59 persons, with a proposed bid of 1000 FF. It was applied by EdF.
ii. A further variant was conducted by EdF asking 61 respondents to give their WTP for an increase in LE of 1 month, 3 months, 12 months.
iii. The risk reduction was stated as in the original questionnaire and then translated to the individual LE gain (calculated by the computer in response to age and gender of the individual); this version was applied to 52 persons, questioned on their WTP (open question only) for a 5/1000 risk reduction for the next 10 years, and between 70 and 80, and followed by in-depth face-to-face debriefing and small group discussion with 3 to 4 respondents.

The LE gain was calculated by Rabl [2002]. For a 5/1000 risk reduction it ranges from 19 to 64 days, depending on sex and age, for a 1/1000 risk reduction it ranges from 4 to 13 days, and for a 5/1000 reduction from 70 and 80 it ranges from 22 to 24 days.

Table C.12. Mean WTP for a gain in life expectancy.

Risk reduction Sequence	1/1000	5/1000	5/1000 from 70 to 80
EDF S1 (version i, 30 respondents) (% of zeros)	68 € (87 %)	169 € (53 %)	76 € (70 %)
EDF S2 (version i, 29 respondents) (% of zeros)	259 € (48 %)	332 € (35 %)	293 € (39 %)
Open question version iii (52 respondents) (% of zeros)		210 € (23 %)	112 € (46%)

S1 : 5/1000 followed by 1/1000
S2 : 1/1000 followed by 5/1000

The results of these two versions, see Table C.12, are not comparable because in the EdF version the LE gain is put in perspective by telling each respondent his/her life expectancy, without any reference to the corresponding risk reduction. In the second version only one change was added to the basic questionnaire: the LE gain corresponding to the risk reduction. In the EdF version people shift from "the risk of death" of the original questionnaire to "a longer life". This affects clearly the answers by increasing the number of zeros. Moreover in the EdF version people express more doubts about the quality of life for the few days gained.

This is reflected in the reasons given for a zero WTP during the debriefing. By a large majority (73% of the zero WTP) the respondents refuse to pay because they consider the gain too short, and not worth taking a medication during 10 years. But they also said (10 % of the zero WTP) that they would agree to pay something if they were sure to enjoy a better quality of life during that extra period.

It is instructive to explain the difference in values between the sequences S1 and S2. For S1 (first LE gain corresponding to 5/1000 then LE gain corresponding to 1/1000) more persons gave a zero (53%) for the first LE gain, and those with a positive WTP lowered (as expected) the second and third LE gain. For S2 (first LE gain corresponding to 1/1000 then LE gain corresponding to 5/1000) the number of zeros was smaller for the first LE gain (48%), and for the second and third LE gain the majority gave the same WTP (12 of the 19 with positive WTP). We can infer that people have difficulty distinguishing small LE gains, and they will more readily decrease a WTP when the gain is reduced than increase it when the gain is increased. This asymmetry arises because the first WTP is often already close to the maximum people are willing to pay: they are reluctant to give more, whereas a decrease in response to a lower LE gain is easier.

For version (ii), WTP increases with LE: 140 €, 234 €, 377 €, as expected.

A closer look at the responses to these variants shows that the better the scenario is understood and accepted, the lower are the highest individual WTPs. In the original version the highest WTP for the open question and the 5/1000 reduction was about 6000 €. When the bids were removed and only the open question was asked, the highest WTP was 3650 €. In the life expectancy variants it falls to 1800 € for version (i) and 1500 € for version (iii).

What do people express by stating a WTP: lessons from debriefing and verbal protocol

Debriefing following the variant Open Question
50 respondents, sequence S2 only (first 1/1000 then 5/1000)

Reasons given for zero WTP (in the order of frequency):

- i risk reduction too low (by a large majority);
- ii no certainty that this medication will do what is promised ("a medication not reimbursed cannot be considered as effective", "I don't want to take medication during 10 years, because of side effects");
- iii low income during retirement. Note that Frenchmen retire relatively early, many even well before the mandatory retirement age of 60 for a woman and 65 for a man; in the very large public sector the majority retire at 57.5. So the percentage of respondents with part or all of the payment period during retirement is large, much larger than in the USA or Canada.

How do the respondents with a positive WTP construct their value?

a) If they are in good health (the majority of the sample) those who currently take a daily medication, such as vitamins, medication to reduce hypertension or medication to reduce cholesterol, think of a similar treatment and give small values (500 to 3000 FF); the others, comparable in number, think of an ordinary expense (meal in a restaurant, price of cigarettes, or a present for their grand children) and give values from 100 to 4000 FF, depending on the type of expense used as reference.
b) If they have had cancer or other serious illness they give a higher value (5000 FF), or they refuse to pay anything because they think that their chance of surviving the next 10 years is too low.

Who states the highest values?
A woman of 51, suffering from migraine, gives 12000 FF but is not sure to be able to pay for 10 years. A man of 57, thinking that this questionnaire is an incitation to be generous and to pay for oneself as well as for others, gives 10000 FF for each of the three risk reductions. A man of 65, expresses a kind of fear of death, saying "if I refuse to pay I will die", and gives 10000 FF. The third highest value is 7000 FF, by a man of 44 who refers to his annual cigarette consumption.

7 (20 %) correct their answer mainly to become more consistent, staying in the same range of values (except for one person).

Debriefing and Discussions Following the Variant Life Expectancy
Version (iii) 52 respondents, WTP for the 5/1000 reduction.

Adding information on gain of LE has the following effects on the stated WTP:

1 The range of values decreases, the highest now being 1500 € (note that at the time of this variant the use of the Euro was so well established that we decided to ask for bids in FF and in €). But the number of zeros (25%) does not increase.
2 Women in their fifties express more easily their fear of not having enough income when they or their husband retire, or if their husband dies.
3 Respondents express also their difficulty to state a value and are more inclined to give a lump sum, explaining that it is “a donation” or “to ward off ill fortune” or “for one’s pleasure”.
4 Now only one person corrected the values stated initially, by contrast to the 20% who made corrections in the variant Open Question.

Comments made:
All remarked that the gain of LE was too little; they thought it would be 1 to 5 years (shorter for older, longer for younger people), not just a few weeks.
A typical comment is “take a medication every day, during 10 years, for such a short gain in life?”. “And what will be the quality of life during these few extra days?”.
They acknowledge the importance of giving both pieces of information because : “5/1000, what does that have to do with reality?”
The relation to a daily expense is also mentioned (cigarettes, or a medication they take).

Reasons for zero WTP are:
“It is better to have a healthy way of life than to take a medication during 10 years” or “I prefer not take a medication every day for 10 years”;
“My income is not enough to pay for a medication during 10 years”;
“I don’t want to pay during 10 years to gain only 24 days, with certainly a bad quality of life”;
“I will be dead in 10 years” (persons suffering from cancer or aids).

Conclusions suggested by these comments:

1 Informing respondents about their gain of LE for each risk reduction made the questionnaire much more clear and understandable. They feel that they learn something and find it more interesting and satisfying to respond.
2 They find that the gain of LE is very short. By giving them this new information there were inclined to think in term of gain in years of life, and spontaneously they stated a longer time than the time offered. It was difficult for them to think in term of average. They considered the indicated average LE gain as their own even though we mentioned the wide range of unpredictable possibilities for the individual (from some days to ten years).
3 They question more easily the quality of life: what will be the quality of life during this additional period of my life? This lack of information on the quality of

life is a crucial problem with the current questionnaire, especially the variant Life Expectancy: people tend to associate old age with a bad quality of life.

4 To value an unfamiliar good people need to find a "reference value" in their mind. This reference affects the stated WTP (if the reference is a regular treatment by a doctor it does not generate the same WTP as if the reference value is a vitamin supplement). The most difficult is to control this reference value. But it is important to have an idea about this value because it states a kind of base line of valuation. This point also raises doubts about the transferability of results from one country to another, for example from France (where a scan for osteoporosis costs only 20 €) to the USA (where it costs about $300).

Value of Statistical Life and Value of a Life Year

An analysis of the relation between the WTPs for the risk reduction during the next ten years and the one between 70 and 80 shows that the implied discount rate is very low in France, approximately 2%. Note that this rate is the combined effect of the true discount rate and the evolution of the value of the risk reduction in the future (due to change of income, and due to the perception of the survival probability and of the severity of a future risk compared to one at present). These effects cannot be distinguished with the available data, but there is really no need to do so: for policy applications only this combined effect matters.

In any case, the result for the discount rate is very uncertain in view of all the above mentioned problems with the questionnaire. Since the time period of ten years is not very long and the discount rate low, we neglect discounting altogether. Without discounting the value of statistical life (VSL) is obtained from the WTP for a risk reduction ΔR according to the simple formula

$$\text{VSL} = \text{WTP (for } \Delta R) \text{ €/yr} \times 10 \text{ yr}/(\Delta R) \quad . \tag{1}$$

The corresponding value of a life year (VOLY) is obtained from the WTP and the gain in life expectancy ΔLE as

$$\text{VOLY} = \text{WTP (for } \Delta R) \text{ €/yr} \times 10 \text{ yr}/\Delta LE \text{ (for } \Delta R) \tag{2}$$

where ΔLE = mean gain in life expectancy for the risk reduction ΔR. In view of all the uncertainties we make the approximation of taking the average WTP and the average ΔLE (approximately 40 days or 0.1 yr for $\Delta R = 5/1000$), rather than first calculating the ratios of WTP and ΔLE for each age and gender and then averaging. Results are shown in Table C.13.

Table C.13. WTP, from Tables C.8 and C.10 – C.12, and VSL and VOLY. Only results for the merged sequences S1+S2 are shown, except for the variants Open Question (only S2) and Life Expectancy (only 5/1000 risk reduction).

Version	ΔR	ΔLE, yr	WTP, €	VSL, M€	VOLY, K€
Original questionnaire					
Mean of open question	1/1000	0.02	412	4.12	206
Logit of bids	1/1000	0.02	294	2.94	147
Spike	1/1000	0.02	440	4.40	220
Non-param. of bids	1/1000	0.02	208	2.08	104
Mean of open question	5/1000	0.1	563	1.13	56.3
Logit of bids	5/1000	0.1	482	0.96	48.2
Spike of bids	5/1000	0.1	712	1.42	71.2
Non-param. of bids	5/1000	0.1	287	0.57	28.7
Variants					
Public health	1/1000	0.02	180	1.80	90
Public health	5/1000	0.1	283	0.57	28.3
Open question	1/1000	0.02	196	1.96	98
Open question	5/1000	0.1	256	0.51	25.6
Life expectancy (version iii)	5/1000	0.1	210	0.42	21

The VSL and VOLY values cover a wide range, spanning about an order of magnitude. Extracting a recommended value is a delicate matter, involving subjective judgment. We tend to place somewhat less weight on the numbers for the 1/1000 risk reduction because people seem to have trouble appreciating such a small reduction. Furthermore, all the WTPs based on bids have a significant anchoring bias which tends to drive the values up. Even though the open question and life expectancy variants are based on small samples only, of 50 respondents each, we take them very seriously because the respondents found it easier to state a meaningful WTP. On balance we recommend a range of VSL values from 0.5 to 3 M€ from the French survey, with a central value of 1 M€; the corresponding VOLY numbers range from 25 to 150 K€ with a central value of 50 K€.

VARIANTS TO KRUPNICK'S QUESTIONNAIRE

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Introduction

Some tests on Krupnick's questionnaire in France and GB have pointed out several weaknesses, so it was decided to conduct variants. The aim of a variant is to change only one parameter and to test its influence on the WTP, all other things being equal.

Two variants were implemented on small samples (about 50 persons) in May and October 2002 at the University of Strasbourg (France). The parameters tested are:
- the scenario : a "public health programme" instead of a "product";
- the formulation of the gain proposed in the evaluation: a "gain in life expectancy" instead of a "reduction in risk of death".

Because of the small size of the samples, it was decided to propose only one initial bid (150 euros) to every interviewee instead of the four possible initial bids used in Krupnick. So, to avoid the intrusion of the starting point bias into the comparison between the variants and Krupnick's questionnaire, the results of the variants will be compared to the sub-sample of the French implementation survey who have had the same initial bid (75 persons).

Variant "public health programme"

Reasons for variant

Some tests on Krupnick's questionnaire in France and in GB have highlighted that the scenario "product" implies several bias.

i) One third of the interviewees had doubts about the product effectiveness and for 20% of them these doubts lower their WTP.
ii) The product proposed is a private good, which leads to assess a private WTP. The European health systems suggest that a public good could be more appropriate. Moreover, a “product” appears far from the solutions for pollution abatement, which could rather be in term of public intervention.
iii) Some interviewees don't accept the product and don't want to pay for it while they would be willing to finance another mean to reduce their risk of death.
iv) Because of the culture of free medical treatment in Europe many respondents have difficulties in understanding health actions in terms of their costs.

Implementation

Economic theory suggests that the scenario should be the most plausible i.e. what would be implemented in reality. So a "public health programme" appears relevant, even if the interviewees are informed that it will reduce their own risk of death. In the variant, a "public health programme" replaces the "product" and the description of the "public health programme" is adapted. In this case, the appropriate payment vehicle used is a "financial participation collected by a public agency" to avoid the risk of rejection of tax and the doubt about the final use of money. Table C.14 shows the results in term of WTP.

Table C.14: Mean WTP in euros

Reduction in the risk of death evaluated ↓	French implementation of the Krupnick questionnaire		French implementation Sub-sample "150€"		Variant "public health programme"	
	1st reduction in risk proposed		1st reduction in risk proposed		1st reduction in risk proposed	
	5/1000	1/1000	5/1000	1/1000	5/1000	1/1000
5/1000	837 € (151)	467 € (148)	1203 € (39)	1580 € (36)	327 € (26)	239 € (26)
1/1000	767 € (151)	404 € (148)	821 € (39)	486 € (36)	140 € (26)	220 € (26)

The figures in brackets is the size of the sample

It occurs that the mean WTP from the variant are lower than those from the French implementation of the Krupnick et. al. questionnaire whatever the reduction in the risk of death is (even when the results of the sub-sample are used). So the first conclusion is that the definition of the scenario has an influence on the WTP.

The credibility of the scenario

The credibility of the valuation medium can be assessed through the percentage of interviewees who had doubts about the product, respectively the public health programme, effectiveness (see Table C.15).

Table C.15: Percentage of interviewees who had doubts about the product / public health programme effectiveness

	French implementation Sub-sample "150€"	Variant "public health programme"
People who have doubts	43 %	17 %

From these results, it appears that people better subscribe to the scenario when it is defined as a public health programme.

Acceptability of the payment vehicle

With public financing, there is a risk of bias in the valuation procedure :

- Interviewees have in mind the sum they are willing to finance public goods. This belief could result in a large percentage of lump sum.
- Interviewees can reject the payment vehicle, which could result in a large percentage of null WTP.

Table C.16: Percentage of interviewees who gave a lump sum and a null WTP

	French implementation Sub-sample "150€"		Variant "public health programme"	
	1st reduction in risk proposed		1st reduction in risk proposed	
	5/1000	1/1000	5/1000	1/1000
Lump sum	43 %	55 %	19 %	50 %
Null WTP	10 %	22 %	8 %	15 %

From these results, it appears that people better accept the scenario when it is defined as a public health programme. But from the debriefings of the variant, it appears that 45 % of the null WTP are motivated by the refusal of an additional payment ("we already pay much").

The percentage of the null WTP is lower in the variant than in the French implementation of Krupnick's questionnaire so it couldn't explain the mean WTP from the former as lower than the mean WTP from the latter.

The problem of strategic behaviour in the scenario

With a public good there is also a risk to encounter two sorts of strategic behaviour :

- If interviewees believe payment of their WTP will only be collected from them and/or the benefits will have to be shared with every body, they could give a low WTP in order to avoid to pay for all (free-riding);
- If respondents believe payment could be collected from others, they could refuse to pay.

Sharing the benefits with everybody

We can assess the strategic behaviour in analysing the percentage of interviewees who considered other benefits to the product/public health programme (see Table C.17).

Table C.17: Percentage of people who have thought about other benefits and the influence on their WTP

	French implementation Sub-sample "150€"	Variant "public health programme"
People who have thought about other benefits	**27 %**	**60 %**
Of which :		
- Improvement of your own quality of life	**45 %**	10 %
- Improvement of the quality of life of your relatives	**30 %**	23 %
- Benefits for the overall society	25 %	**67 %**
People who have thought these considerations influence their WTP:		
lower	5 %	**29 %**
raise	**35 %**	10 %
no influence	60 %	61 %

From these results, it appears that :

- in the French implementation of Krupnick's questionnaire, the respondents who have thought about other benefits have also thought about an improvement of their own quality of life (consistent with a private good) ;
- in the variant, in contrary the respondents who have thought about other benefits have thought about benefits for the overall society (consistent with a public good). This finding might be interpreted to mean that these respondents are altruistic.

But, the influence of this consideration on the WTP highlights this is not the case. In fact, among the individuals who have thought about other benefits:

- 29% have given a lower WTP in the case of the public health programme ;
- 35% have given a higher WTP in the case of a product.

In the variant, not only are the respondents not altruistic (upward influence on the WTP for only 10% of them) but they are free riders in refusing to pay for others (lower influence for 29% of them).

The opposite influence on the WTP of respondents who have thought about other benefits could contribute to explain that the mean WTP from the variant is lower than the mean WTP form the French implementation of the Krupnick's questionnaire.

Public good financing
In the variant, interviewees were asked about the best means in which to finance the public health programme (see Table C-18).

Table C.18: Percentage of the 1st choice for the means in which to finance the public health programme

Different means proposed	*1st choice (in %age)*
Increase in social contributions	*22 %*
Increase in indirect taxes	*14 %*
Increase in direct taxes (cigarettes, alcohol, other products)	*65 %*
Other	*0 %*

It can be noticed that no one has proposed another means, which can be taken to suppose that no one has rejected public financing.
Nevertheless, it is important to note that among respondents who have chosen an increase in direct taxes, of which those on cigarettes, 80% are non-smokers. Also note that one third of these individuals have given a lower WTP than the mean WTP. We can assume that these respondents had a strategic behaviour in relying on others to finance the public health programme.

Variant "life expectancy"

Reasons for variant
In the Krupnick et. al. questionnaire, interviewees are asked their WTP to reduce their risk of death by 1/1000 and 5/1000 per year over the next ten years. This choice raises two problems:

i) Some respondents do not understand the probability concept and/or do not conceptualise the reduction proposed. Despite the explanations given in the questionnaire, 10% said they had not understood the risk concept.
ii) The aim of this work package is to assess the value of life years (VOLY) on an empirical basis. Krupnick's questionnaire does not allow this directly: it assesses the value of a statistical life (VSL). The VOLY can be estimated indirectly from VSL and the transformation of the reduction in the risk of death into life expectancy lost. This calculation assumes that respondents are able to equate the two measures.

Two variants were designed :
- EV1 : life expectancy gains exactly correspond to the reduction in the risk of death by age and by sex. Ari Rabl made the calculation.
- EV2 : 1 month, 3 months and one year of life expectancy gains are proposed to all interviewees. One month and 3 month correspond respectively to the lower and the upper bonds of a reduction in the risk of death by 5/1000 per year over the next 10 years used in Krupnick's questionnaire.

The objective is to replace the reductions in the risk of death used in Krupnick et. al. by gains in life expectancy. The explanations are changed in accordance with this new concept. Table C.19 shows the results in term of WTP.

Table C.19: Mean WTP in €

Reduction in the risk of death / life expectancy gains evaluated ↓	French implementation of Krupnick's questionnaire		French implementation Sub-sample "150€"		Variant "EV1"		Variant "EV2"
	1st reduction in risk proposed		1st reduction in risk proposed		1st reduction in risk proposed		-
	5/1000	1/1000	5/1000	1/1000	5/1000	1/1000	
5/1000	837 € (151)	467 € (148)	1203 € (39)	1580 € (36)	149 € (29)	332 € (29)	
1/1000	767 € (151)	404 € (148)	821 € (39)	486 € (36)	46 € (29)	259 € (29)	
1 month	-						140 € (61)
3 months	-						234 € (61)
1 year	-						377 € (61)

The figures in brackets indicate the size of the sample

It appears that the mean WTP from the variants are lower than those from the French implementation of Krupnick's questionnaire (even when the results of the sub-sample are used). So the first conclusion is that the expression of the reductions in the risk of death into life expectancy gains has a significant influence on the WTP.

The understanding of the "risk of death" / "life expectancy" concept could be assessed by the rate of respondents who told they have understood it and the rate of mistake to the comprehension test (see Table C.20).

Table C.20: "Risk of death" / "Life expectancy" understanding

	French implementation Sub-sample "150€" (risk)	Variant "public health programme" (risk)	Variant "EV1" (life expectancy)	Variant "EV2" (life expectancy)
Good understanding*	82%	78.5%	91.5%	88.5%
Made a mistake to the 1st comprehension test	23%	25%	7%	5%

* Understanding quoted to 4 and 5 on a scale of 5

From these results, it appears that the "life expectancy" concept is better understood by respondents than the "risk of death" one. Higher rates of null WTP in the variants than in the French implementation of Krupnick's questionnaire could explain why the mean WTP from the formers are lower than mean WTP from the latter.

Table C.21: Percentage of zero WTP

Reduction / gain	French implementation Sub-sample "150€"	Variant "EV1"	Variant "EV2"
1	25 %	51 %	54 %
2	24 %	61 %	46 %
3	-		26 %

1 : the 1st proposed reduction / gain i.e. 1/1000 or 5/1000 over the new 10 years for " French implementation " and the corresponding life expectancy gains for EV1 and 1 month for EV2
2 : the 2nd proposed reduction / gain i.e.1/1000 or 5/1000 over the new 10 years for " French implementation " and the corresponding life expectancy gains for EV1 and 3 months for EV2
3 : the 3rd proposed gain i.e. 12 months for EV2

The debriefings after the variants highlights the null WTP is greatly motivated (more than 70%) by the too small size of the proposed gains relative to the constrain of using the product during 10 years, especially in EV1. It is also interesting to note that the 2nd motivation is the lack of indication about the quality of life.

Compared to a formulation in terms of "risk of death over the next 10 years", the new formulation probably brings interviewees to put the proposed gain at the end of their life. The comparison between the mean WTP for a future (from 70 to 80 years old) reduction of the risk of death proposed in Krupnick's questionnaire and the mean WTP for the life expectancy gains proposed in EV1 and EV2 highlights they are close together.

Table C.22: WTP in €

Reductions in the risk of death or life expectancy gain valued ↓	French implementation: 1st reduction in risk proposed		Variant "EV1": 1st reduction in risk proposed		Variant "EV2"
	5/1000	1/1000	5/1000	1/1000	
5/1000 over the next 10 years	837 € (151)	467 € (148)			-
corresponding life expectancy gain from 19 to 64 days			149 € (29)	332 € (29)	
5/1000 between 70 and80 years old	516 € (199)	368 € (199)			
corresponding life expectancy gain from 22 to 24 days			79 € (17)	293 € (18)	
Life expectancy gain of 1 month (EV2)	-				140 € (61)
Life expectancy gain of 3 months (EV2)					234 € (61)

The figures in brackets indicate the size of the sample

Conclusion and recommendations

The choice of a scenario consistent with mortality valuation is still open. In the variant "public health programme", we test a scenario better adapted to the European public health context that may be more credible than the "product" used in Krupnick. The credibility of the scenario seems to be improved in the variant. In counter part, new strategic behaviour relating to provision of the public good has appeared and could explain the downward influence on the WTP in the variant compared to the WTP from the French implementation of Krupnick's questionnaire. So, further work is need either to elaborate a more appropriate scenario to mortality valuation or to eliminate bias such the strategic behaviour.

Valuation of a life expectancy gain is closer to the epidemiological indicator in an environmental pollution context than a reduction in the risk of death. Moreover, in the variant "life expectancy", the understanding of the risk concept seems to be improved when it is formulated in term of life expectancy. The downward influence on the WTP in the variants compared to the WTP from the French implementation of Krupnick's questionnaire could be explained by :

- respondents found the life expectancy gains too small relative to the constraint of using the product during 10 years and need information about the quality of life – the problem of valuing a small change, whether in life expectancy or in risk remains;
- the "life expectancy" formulation brings them to put the proposed gain at the end of their life – this may be appropriate for acute effects but not necessarily for chronic effects.

Further research is needed in valuing life expectancy. The choice of a scenario consistent with the life expectancy gain length need to be investigated and the variant "public health programme" could be considered as a first step. Considerations about quality of life need to be integrated to characterize the life expectancy gain.

THE WILLINGNESS TO PAY FOR MORTALITY RISK REDUCTIONS: EVIDENCE FROM ITALY

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I. Introduction

The purpose of this document is to summarize the results of a contingent valuation survey eliciting willingness to pay (WTP) for reductions in one's own risk of death. The survey was self-administered using the computer to a sample of respondents recruited at various locations in Italy. The original questionnaire was developed by Krupnick et al. (2001), and was translated into Italian for the purpose of the present study.[9]

The goal of the study is four-fold. First, we wish to see if the WTP for a reduction in the risk of dying depends on age. This is an important policy question, as the majority of the lives saved by environmental policies are those of the elderly. Second, we wish to see if WTP is influenced by a person's health status. Once again, epidemiological evidence suggests that the mortality effects of pollution (and hence the mortality benefits of environmental policies) fall disproportionately on the shoulders of persons with compromised health. Third, we wish to see if it possible to elicit meaningful WTP figures for a latent risk reduction, i.e., a risk reduction that takes place in the future. This is the type of risk reduction delivered by many environmental policies that reduce cancer risks. Fourth and last, by conducting a survey based on the same instrument in different countries, we wish to explore the issue of transferability of WTP figures from one country to another.

The remainder of this report is organized as follows. In section II, I describe the questionnaire. In section III, I summarize sampling procedures and experimental designs. In section IV, I present descriptive statistics for the respondents. In section V, I examine the respondents' comprehension of risks. In section VI,I present WTP figures, and in section VII regression models that test internal validity of the responses to the payment questions. Section VIII describes a pooled data model that uses two observations for each respondent. Section IX examines the WTP for the future risk reduction, and section X proposes future work.

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[9] The questionnaire had previously been administered to samples of Canadians and US residents, a sample of residents in the Bath area in the UK, and a sample of residents of Strasbourg in France. The results of these studies are reported in Krupnick et al. (2001) for Canada, Alberini et al. (forthcoming) for the US, and in Alberini (2002) for the UK. Alberini (2003) reports descriptive statistics and results of preliminary econometric models of WTP based on pooling the data collected in Italy, the UK, and France.

II. Description of the questionnaire

The questionnaire begins by asking respondents questions about their age, own health, and the health of their family members. It then provides a simple tutorial about the concept of probability, which is followed by two questions meant to assess the respondent's comprehension of probability. In what we term the probability quiz, the respondent is shown grids of 1000 squares, with red squares depicting the risk of dying, for person A and person B, and is asked which person has the higher risk of death. The probability choice question has a similar format, but asks the respondent to state which person, A or B, he would rather *be*.

Respondents are then shown their baseline risk of death over the next 10 years, which varies with gender and age, and are given examples of behaviors and medical intervention that reduces the risk of death. They are also given qualitative information about the cost of reducing their own risk of death.

The questions at the heart of this study are, of course, the WTP questions. Respondents are asked to value three risk reductions from their baseline risk of dying over the next ten years:

- (i) a risk reduction of 5 in 1000, to begin immediately and to be incurred over the next 10 years,
- (ii) a risk reduction of 1 in 1000, to begin immediately and to be incurred over the next 10 years, and
- (iii) a risk reduction of 5 in 1000, which would begin at age 70, spread over the following 10 years, and refer to the baseline risk at age 70.

Respondents were asked to think about a product that would deliver the stated risk reduction, and were asked whether they would buy the product at the stated price, with payments to be made every year for 10 years, beginning immediately. The risk reductions to be valued by the respondent are private, and the elicitation technique is dichotomous choice with one dichotomous-choice follow-up question.[10] The bids are shown in Table D.1.

Table D.1. Bid design. All amounts in Euro.

Bid set	Initial bid	If yes	If no
I	80	170	35
II	170	570	80
III	570	830	170
IV	830	1140	570

[10] Respondents who said that they would buy the product for the cost stated to them in the initial payment question were asked whether they would pay a higher amount for the product. Respondents who declined to purchase the good at the initial amount were asked whether they would pay a lower amount. Respondents who answered "yes" to the initial and follow-up payment question were then asked what their maximum willingness to pay was, whereas respondents who declined to pay the initial amount and the amount in the follow-up question were asked whether they were willing to pay anything at all, and, if so, how much.

This was followed by questions about the respondent's demographics and by debriefing questions attempting to gauge the respondent's acceptance of risks and risk reduction scenario. At the end of the survey, subjects were given the Short Form-36 questions to assess their physical and mental health.

III. Mode of Administration and Sampling

Respondents were selected among participants in computer classes at the FEEM's Multimedia Library in Venice, Milan, Turin and Genoa, and from workers of the Milan area for a total of 292 completed interviews.

The objective of the sampling was to obtain a random sample stratified by age classes and gender. The age classes were 40-50 year old, 51-60 and above 60. Approximately one third of the sample was assigned to each stratum with an even split between men and women. Sampling took place in two waves. The first wave involved the collection of the first 155 completed surveys and took place in different cities (Venice, Milan, Genoa and Naples). The sampling method was based on convenience sampling as people who took part into information technology priming classes were asked to fill in the survey in various centers. The second wave was designed to complement the first in terms of achieving the target numbers in each age/gender category, and was carried out in Milan and surroundings by enumerators that carried out surveys on laptops.

IV. Descriptive Statistics of the Respondents

Descriptive statistics are reported in Table D.2. The sampling plan restricted attention to persons older than 40 years of age and specified quotas for the various age groups. The average age in the sample is about 57 years, as is appropriate and consistent with the sampling frame. The oldest age group (ages 70 and older) accounts for about 14 percent of the sample.

The sample is relatively well balanced in terms of gender, with only a slight prevalence of women over men, and the average number of years of schooling ranges is about 13, which corresponds to completion of high school.

Mean and median annual household incomes are reported in both EUR and in PPP US$. The latter figure is obtained through multiplying household income in EUR by 0.813.

Table D.2. Descriptive Statistics of the Respondents in the Italy Study.

Variable	Sample mean or percentage
Age (years)	57.04
Age group 40-49	28.42%
Age group 50-59	33.22%
Age group 60-69	23.97%
Age group 70 and older	14.38%
Education (years of schooling)	12.99
Male	48.63%
Household income in EUR	
Mean	40,115
Median	25,000
Household income in 2002 US $ using PPP:	
Mean	32,613
Median	20,325

Respondents were asked several questions about their own health status. For example, we asked them how they would rate their health, relative to others of the same age. The possible response categories are excellent, very good, good, fair and poor. As shown in Table D.3, about 42 percent of the sample rates their health as excellent or very good when compared with others the same age. This figure is comparable to the corresponding proportion in the France study, but is much lower than the corresponding statistics for the UK, Canada, and the US.

We also asked the respondents whether they suffered from various chronic ailments. As We found that about 15% has heart disease, 13% has a chronic respiratory illness, 33% has high blood pressure, and less than 7% has or has had cancer. Table D.3 also reports the average baseline risk for the sample. In the survey, respondents were told the risk of dying over ten years for the average person of their age and gender. The average baseline risk in the sample is about 50 per 1000, and is thus much lower than the baseline risks for the UK, France, the US and Canada studies.

Table D.3. Health status of the Respondents.

Variable	Sample average or percentage
Rates own health as very good or excellent relative to others same age	42.12 percent
CARDIO	15.41 percent
LUNGS	12.67 percent
PRESSURE (high blood pressure)	33.33 percent
CANC	6.85 percent
Any one of CARDIO, LUNGS, PRESSURE, or a stroke (cancer excluded)	44.86 percent
Baseline risk of dying over the next 10 years	49.94 in 1000

Respondents were also asked to tell us what age they expected to live to. On average, respondents stated that they expected to live to the age of 84 (implying that the average remaining life was 27 years). Subjects were questioned about their subjective probability of surviving to age 70, and the average in the sample was a 46% probability.

V. Risk Comprehension and Acceptance of the Survey

Table D.4 displays the percentages of respondents who failed the probability test, chose the person with the higher probability of dying in the probability choice question, or otherwise report having problem understanding the concept of risk. These figures are comparable to those from the UK and France.

Table D.4. Risk comprehension.

	Percentage of the sample
Wrong answer to the probability quiz	11.64
Confirms wrong answer to the probability quiz	2.74
Probability choice question: -- Prefers person with the higher risk of dying -- indifferent	 11.99 10.96
Confirms wrong answer in the probability quiz and choice question	3.08
Thinks he/she understands probabilities poorly (FLAG6=1)	27.41
FLAG1=1	3.77

In Table D.5, we show the proportions of respondents who, based on their answers to the debriefing questions, might be argued to have rejected or misunderstood certain aspects of the scenario.

Table D.5. Acceptance and comprehension of the scenario.

FLAG and description	**Percentage of the sample**
FLAG7: did not believe the risk figures	21.92
FLAG8: had doubts about product effectiveness	41.10
FLAG9: had doubts about product effectiveness and doubts affected WTP	27.05
FLAG11: has thought about other benefits of the product (or does not know)	38.70
FLAG14: did not understand that the payment would start immediately	9.93
FLAG15: did not consider whether he can afford it	28.77
FLAG16: did not understand payment scheme	5.82

We ran several probit regressions to see if the likelihood of reporting some of these problems with the scenario was related to the respondent age and to other individual characteristics.

Results are mixed. For example, a probit regression of the FLAG7 indicator dummy on three age group dummies, a gender dummy, and a dummy denoting the presence of a chronic illness suggests that persons in the oldest age group and persons with a chronic illness are less likely, and males are more likely to, question the risk figures. However, while the magnitude of the respective coefficients can be large, these effects are statistically significant only at the 10 percent level (see Table D.6).

Table D.6. Determinants of rejection/misunderstanding of the scenario.

Probit equation:

Dep. Variable: flag7 (did not believe the risk figures) N=270

Parameter	DF	Estimate	Standard Error	Wald Chi-Square	Pr > ChiSq
Intercept	1	-0.6046	0.3763	2.5812	0.1081
age5059	1	0.0735	0.2190	0.1127	0.7371
age6069	1	-0.0745	0.2465	0.0914	0.7624
ag70plus	1	-0.6341	0.3586	3.1272	0.0770
chronic	1	-0.3537	0.1911	3.4239	0.0643
male	1	0.2993	0.1807	2.7424	0.0977
educ	1	-0.00425	0.0251	0.0286	0.8657
pcappinc	1	-0.00749	0.00722	1.0771	0.2993

Dep. Variable: flag8 (respondent has doubts about product effectiveness) N=270

Parameter	DF	Estimate	Standard Error	Wald Chi-Square	Pr > ChiSq
Intercept	1	-0.0144	0.3350	0.0019	0.9656
age5059	1	0.2064	0.2000	1.0655	0.3020
age6069	1	-0.1190	0.2204	0.2915	0.5893
ag70plus	1	-0.5153	0.2870	3.2231	0.0726
chronic	1	0.0974	0.1672	0.3390	0.5604
male	1	-0.3246	0.1589	4.1714	0.0411
educ	1	-0.00819	0.0223	0.1346	0.7137
pcappinc	1	-0.00019	0.00582	0.0011	0.9740

Definition of variables: age5059=dummy equal to one if the respondent's age is between 50 and 59 years; age6069=dummy equal to one if the respondent's age is between 60 and 69 years; ag70plus=dummy equal to one if the respondent's age is 70 years or older; chronic=see Table D.3; educ=years of schooling; pcappinc=household income/household size.

When we fit a similar model to FLAG8, a dummy indicator equal to one if the respondent has doubts about the effectiveness of the product (and would thus question the risk reduction to be valued), we found that subjects in the oldest age group and males were less likely to question the effectiveness of the product.

We did not find any association between FLAG11, the dummy denoting that the respondent has thought of other benefits of the product, and FLAG16 (which is equal to 1 if the respondent did not understand the payment scheme) and the individual characteristics of the respondent. We did, however, found that the likelihood of refusing to buy the product and hence failing to consider if it is affordable (FLAG15) is affected solely by the respondent's education level, more highly educated people being more likely to refuse the product.

VI. Responses to the Payment Questions and WTP Figures

The percentages of 'yes' responses to the initial payment questions for the 5 in 1000 and 1 in 1000 risk reductions are displayed in Figure D.1.

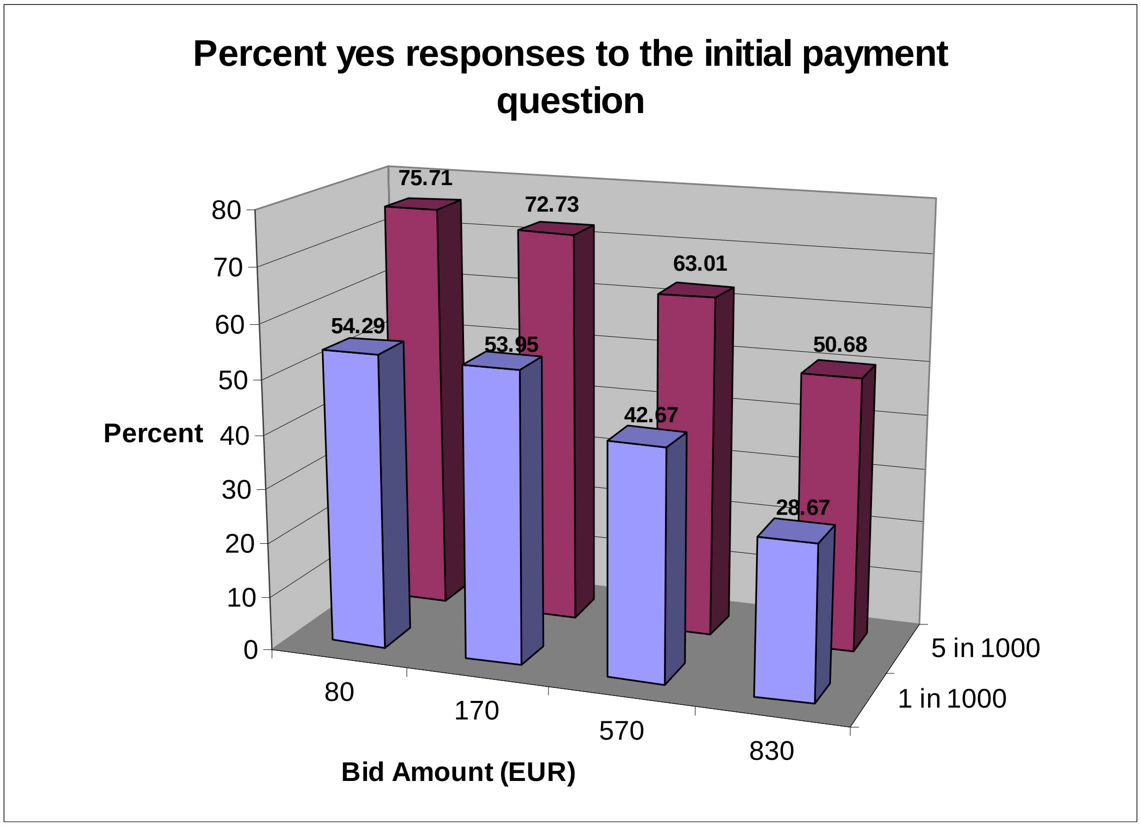


Figure D.1. Percentages of "yes" responses to initial payment questions (risk reductions of 5 in 1000 and 1 in 1000)

Three findings emerge from this figure. First, the proportion of 'yes' responses to the initial payment question declines with the bid amount for both risk reductions, implying that the responses are consistent with economic theory. Second, median WTP is roughly 830 EUR for the 5 in 1000 risk reduction, and between 170 and 570 for the 1 in 1000 risk reduction. Third, for the 5 in 1000 risk reduction the bid amounts are placed to the left of the median, implying that we cannot nail down the upper tail of the distribution of WTP. We would therefore expect inefficient, possibly unstable estimates of mean WTP if we assume that the distribution of WTP is skewed, as is the case with the Weibull and lognormal.

When we combine the responses to the initial and follow-up payment questions for the 5 in 1000 risk reduction, we notice that there is a prevalence of yes-yes responses (53.08% of the sample), followed by no-no responses (28.08 percent of the sample), and that yes-no and no-yes account for a smaller proportion of the observations (12.33 and 6.51%, respectively). Our subjects are, however, willing to pay much less for the smaller risk reduction, as is confirmed by the fact that no-no responses become the category with the highest frequency (43.15%), followed by the yes-yes responses (32.19%), whereas yes-no and no-yes answers account for a similar percentage of the observations (about 12 percent).

Further inspection of the data reveals that 28.08% of the respondent is not willing to pay anything at all for the 5 in 1000 risk reduction, and that 43.15% would not pay anything for the 1 in 1000 risk reduction. These figures are broadly consistent with the notion that people hold lower WTP amounts for the smaller of the two risk reductions.

To obtain estimates of mean and median WTP, we combine the responses to the initial and follow-up payment questions to form intervals around the respondent's (unobserved) WTP amount. For example, if a respondent is willing to pay the initial bid of, say, 170 EUR, and declines to pay the follow-up amount of 570 EUR, it is assumed that his WTP falls between 170 and 570 EUR. We further assume that WTP follows the Weibull distribution with scale parameter σ and shape θ, and estimate these parameters using the method of maximum likelihood. The log likelihood function of the WTP data is:

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

where WTP^L and WTP^U are the lower and upper bound of the interval around the respondent's WTP amount. Equation (1) describes an interval-data model. Separate interval-data models are fit for the immediate 5 in 1000 and 1 in 1000 risk reductions.

Equation (1) can be replaced with the corresponding expressions for the lognormal and for other distributions. As shown in Table D.7, we experimented with lognormal, loglogistic and exponential distributions, but found that, based on the Akaike criteria (which is equal to the log likelihood function, minus the number of parameters to be estimated), the Weibull always had a better fit.[11]

[11] Distributions like the normal and logistic are not considered for two reasons. First, they admit negative values, which is not acceptable in this context (people should hold positive WTP amounts for a reduction in risk). Second, their fit is much worse than that of any of the skewed distributions shown in table 7.

Table D.7. Goodness of fit of various distributions of WTP (based on interval-data models).

5 in 1000 risk reduction

	Weibull	Log normal	Log logistic	Exponential
Log likelihood	-298.72	-300.03	-299.85	-303.26
Number of parameters to be estimated	2	2	2	1

1 in 1000 risk reduction

	Weibull	Log normal	Log logistic	Exponential
Log likelihood	-322.66	-323.32	-324.16	-333.26
Number of parameters to be estimated	2	2	2	1

Accordingly, in what follows we work with the Weibull distribution. In addition to goodness-of-fit considerations, another reason for preferring the Weibull distribution is that in our experience the Weibull has generally been better-behaved than the other positively skewed distributions here examined. The Weibull and the other distributions generally agree in terms of their estimates of median WTP, but may produce very different figures for mean WTP. In addition, the Weibull distribution has a flexible shape: Depending on the value of the shape parameter theta, the density of the Weibull variate can be positively skewed (for theta between 0 and 3.6), symmetric (for theta approximately equal to 3.6), and even negatively skewed (for theta greater than 3.6).

The mean of a Weibull variate is equal to:

(2) $$\sigma \cdot \Gamma\left(\frac{1}{\theta}+1\right)$$

while median WTP is equal to:

(3) $$\sigma \cdot \left[-\ln(0.5)\right]^{1/\theta}.$$

With WTP, experience suggests that mean WTP tends to be two or even three times as large as median WTP. We regard median WTP as a conservative, but robust and more reliable, estimate. For this reason, we report median WTP figures for the 5 in 1000 risk reduction in Table D.8a below.

Table D.8a. Mean and Median annual WTP for the risk reductions beginning now. Interval-data Weibull model. Complete sample. (Standard errors in parentheses.)

	5 in1000 risk reduction over the next 10 years	1 in 1000 risk reduction over the next 10 years
Mean WTP (EUR)	1448 (326)	698 (107)
Median WTP (EUR)	724 (86)	309 (36)
Median WTP after PPP conversion to 2002 US $	586.44 (71.38)	251.22 (29.27)

Table D.8b. Mean and Median WTP for the risk reductions beginning now. Estimates are based on a double bounded non-parametric WTP estimator (Turnbull-Kaplan-Meier). Full sample. Standard errors in parentheses.

	5 in1000 risk reduction over the next 10 years	1 in 1000 risk reduction over the next 10 years
Mean WTP (EUR)	470 (39)	274 (30)
Median WTP (EUR)	170 (47)	80 (49)
Median WTP after PPP conversion to 2002 US $	141.46 (38.43)	65.04 (39.84)

Table D.8a shows that mean WTP is twice median WTP, both in the case of the 5 in 1000 and the 1 in 1000 risk reductions.

An important question is whether WTP is sensitive to scope (see Hammitt and Graham, 1999). A Wald statistic of 4.76 results in the rejection of the null that mean WTP is the same across the 5 in 1000 and 1 in 1000 risk reductions. The Wald statistic of the null that median WTP is the same across the two risk reductions is 19.80. We therefore reject the null hypothesis at all the conventional significance levels. We remind the reader that each of these Wald test is distributed as a chi square with one degree of freedom under the null, and that the critical limit at the 5% significance level is 3.84. We also remind the reader that the two Wald tests refer to an *internal* scope test, and that they are based on the assumption that an individual subject's WTP amounts for the two risk reductions are independent of one another.

While our estimates of WTP are sensitive to scope, they are not strictly proportional to the size of the risk reduction. Assuming, once again, that the WTP responses are independent across the two risk reductions, even within a respondent, Wald tests reject the null hypotheses that mean (median) WTP for the 5 in 1000 risk reduction is 5 times that for the 1 in 1000 risk reduction. The Wald statistic is 10.67 for mean WTP and 16.83 for median WTP, each falling in the rejection region of the chi square with one degree of freedom at all the conventional significance levels.

Mean WTP for the 5 in 1000 risk reduction is about twice mean WTP for the 1 in 1000 risk reduction. Median WTP for the 5 in 1000 risk reduction is 2.34 times median WTP for the 1 in 1000 risk reduction.

To compute the VSL implied by these WTP figures, we simply divide WTP by the annual risk reduction, assuming that the risk reduction over the course of 10 years would be accrued uniformly.[12] We have a total of four possible VSL values (one for each of mean and median WTP, and one for each of the two risk reductions). The VSL ranges from 1,448,000 to 2,896,000 EUR.

The WTP figures from the fully parametric approach may be compared with estimates of mean and median WTP from a non-parametric procedure (the Turnbull-Peto variant of the non-parametric Kaplan-Meier estimator), which are reported in Table D.8b.[13] These estimates are based on a conservative interpretation of the WTP responses, and tend therefore to be lower than those shown in Table D.8b. Mean and median WTP for the 5 in 1000 risk reduction are 470 and 170 EUR, whereas the same welfare statistics for the 1 in 1000 risk reduction are 274 and 80 EUR. The non-parametric approach confirms that WTP for the larger risk reduction is greater than that for the smaller risk reduction, but WTP is not strictly proportional to the size of the risk reduction, as the former figures are about twice as large as the latter.

VII. Internal Validity Tests

To check internal validity, we relate WTP to covariates using an accelerated life Weibull model. Specifically, we allow the scale parameter of the Weibull to vary across individuals as a function of variables thought to be associated with willingness to pay:

$$\sigma_i = \exp(\mathbf{x}_i \boldsymbol{\beta}), \tag{4}$$

where $\mathbf{x}_i$ is a 1×p vector of regressors, and β is a p×1 vectors of coefficients. This is equivalent to specifiying the equation:

$$\log WTP = \mathbf{x}_i \boldsymbol{\beta} + \varepsilon_i, \tag{5}$$

where ε follows the type I extreme value distribution with scale θ.

As before, we form intervals around the respondent's true WTP amount by combining to the responses to the initial payment questions and the first follow-up questions, and ignore the responses to subsequent follow-ups.

[12] Alternatively, one could sum the discounted annual payments, but the latter approach would require making assumptions about the discount rate.

[13] See Haab Timothy C., and Kenneth E. McConnell (1997), "Referendum Models and Negative WTP: Alternative Solutions," *Journal of Environmental Economics and Management*, 32(2):251-270.

Results for several specifications are shown in Tables D.9 and D.10 for a sample that excludes respondents with FLAG1=1 from the sample. These are the subjects that gave the wrong answer in the probability test and chose the person with the higher risk of dying in the probability choice question. This cleaned sample is comprised of 281 individuals.

We wish to explore the impact of several factors on WTP. First, economic theory predicts that WTP should, all else the same, be increasing in baseline risk. However, an interval-data regression of WTP on baseline risk (not reported) yields a negative and significant coefficient on risk.

Willingness to pay should also increase with the size of the risk reduction, and indeed in the previous section we conducted scope tests. In Table D.9, we report the results of an interval-data regression of WTP on the proportional risk reduction experienced by the respondents, i.e., 5 in 1000 divided by baseline risk. (The equation refers to WTP for the 5 in 1000 risk reduction.) The table shows that WTP increases significantly with the size of the proportional risk reduction. It should be noted that the proportional risk reduction tends to be larger in this study than for the US, Canada, France and UK samples, because baseline risk is very small in this group.

Additional WTP regressions are displayed in Table D.10. The specification of column (A) in Table D.10 wishes to answer our first basic question: Does WTP for an immediate risk reduction depend on age? To answer this question, we created dummies indicating whether the respondent belongs to the 50-59, 60-69 and 70-year old or older age groups. As shown in column (A) of the table, WTP is indeed lower in the oldest age group. Specifically, persons of age 70 and older hold WTP amounts that are about 61% lower than those of the reference group (40-49 year-olds).

In specification (B), we control for age, and add other regressors such as the gender dummy, years of schooling (EDUC), and the logarithmic transformation of income per family member. The coefficient on the latter is the income elasticity of WTP, which is assumed to be constant over all the entire range of income values. As shown in the table, the coefficient on the age 70 and older is now a bit lower and is now statistically significant only at the 10% level. The coefficient on the MALE dummy is -0.40, suggesting that males are willing to pay about 33 percent less than females, all else the same. The coefficient on years of schooling is positive, but insignificant, and the income elasticity of WTP is about 0.27. This figure is consistent with estimates from other studies about mortality risk reductions.

Table D.9. WTP regressions for the 5 in 1000 risk reduction. Interval-data model.

```
                                The LIFEREG Procedure

                                  Model Information

                         Data Set                     WORK.ITALY1
                         Dependent Variable           Log(dblow5)
                         Dependent Variable          Log(dbhigh5)
                         Number of Observations               281
                         Noncensored Values                     0
                         Right Censored Values                145
                         Left Censored Values                  81
                         Interval Censored Values              55
                         Name of Distribution             Weibull
                         Log Likelihood              -287.7741718


           Algorithm converged.


                             Type III Analysis of Effects

                                                  Wald
                     Effect           DF    Chi-Square    Pr > ChiSq

                     propreduction     1        4.8423        0.0278


                            Analysis of Parameter Estimates

                                   Standard   95% Confidence     Chi-
             Parameter     DF Estimate    Error       Limits       Square Pr > ChiSq

             Intercept      1   6.6832   0.1820   6.3265   7.0400 1348.07     <.0001
             propreduction  1   1.1576   0.5261   0.1266   2.1887    4.84     0.0278

propreduction=5 in 1000/baseline risk.
```

In column (C), we add two health status dummies: CHRONIC, which takes on a value of one of the respondent has any cardiovascular or respiratory illness, and a dummy for whether he has had or currently has cancer. Neither coefficient is statistically significant, although it should be noted that the coefficient on CANCER is quite large and positive, whereas that on CHRONIC is very small and negative.

We experimented with using other measures of the respondent's health status, but to no avail. Using the individual dummies for cardiovascular problems (CARDIO), chronic respiratory illness (LUNG), high blood pressure (PRESSURE) in lieu of the aggregate CARDIO resulted in insignificant coefficient, and a similar result was noted when we dropped these four dummies and replaced them with a binary indicator (GOODHEAL) for whether the respondent thought that his or her health was excellent or very good relative to others of the same age.

No significant associations are found when serious concerns about the respondent's health status are measured by whether he or she has been admitted to the hospital or has reported to the emergency room for a cardiovascular or respiratory problem in the last five years, nor when we include in the regression a dummy for chronic illnesses or cancer among family members (regression not reported).

Table D.10. WTP for the 5 in 1000 risk reduction. Cleaned sample. T statistics in parentheses.

	(A) Effect of age N=281	(B) Age and individual characteristics N=270	(C) Age, individual characteristics and health status N=270
Intercept	7.3162*** (28.70)	6.5788*** (11.96)	6.5355*** (11.66)
Age 50 to 59	-0.3895 (-1.25)	-0.3933 (-1.20)	-0.4113 (-1.24)
Age 60 to 69	-0.1132 (-0.33)	-0.0707 (-0.20)	-0.0491 (-0.14)
Age 70 and older	-0.9444*** (-2.59)	-0.7870* (-1.91)	-0.7842* (-1.80)
Male		-0.4045 (-1.64)	-0.3713 (-1.50)
EDUC		0.0226 (0.63)	0.0254 (0.70)
Log income per household member		0.2665* (1.66)	0.2532 (1.58)
CHRONIC			-0.0459 (-0.17)
Cancer			0.6266 (1.17)
Log likelih.	-286.57	-272.35	-271.54

* = significant at the 10% level. ** = significant at the 5% level. ***= significant at the 1% level.

Because the data were collected in different cities, we re-ran the regressions of Table D.10 after city dummies were included. The results from two such regressions, comparable to the specifications of columns (B) and (C) of Table D.10, are reported in the end of this appendix (Tables D.14 and D.15). Controlling for the city of residence of the respondent implies that WTP is systematically larger for subjects living in Turin, Genoa, Venice and Milan than it is for Naples residents. The coefficients on the other regressors, however, are very close to the corresponding estimates from models without the city dummies.[14] In particular, the age effects are virtually the same and the lack of any association between WTP and the health status of the respondent is confirmed in the regressions with city dummies.

We ran these regressions using samples constructed following other data cleaning criteria. Unfortunately, we cannot claim that our results are robust to changing the data cleaning criteria. For example, when we exclude respondents with FLAG4=1 or

[14] We also checked whether those respondents who took the survey at the FEEM multimedia library in Milan were systematically different in their responses to the WTP questions from those respondents who self-administered the questionnaire at Milan sites, but found that, once one controls for age, income, gender and education level, the coefficients on the dummies for the Milan survey locale were not statistically different from one another.

FLAG5=1 (respondents who confirmed the wrong answer to the probability test or confirmed that they prefer to be the person with the higher risk of death[15]) the coefficients on the regressors are no longer significant.

Table D.11. WTP for 1 in 1000 risk reduction.

Interval data model for willingness to pay for 1 in 1000 risk reduction. N=270.

Analysis of Parameter Estimates

Parameter	DF	Estimate	Standard Error	95% Confidence Limits		Chi-Square	Pr > ChiSq
Intercept	1	5.7835	0.5213	4.7617	6.8052	123.08	<.0001
age5059	1	-0.4827	0.3013	-1.0733	0.1079	2.57	0.1092
age6069	1	0.2036	0.3270	-0.4373	0.8445	0.39	0.5335
ag70plus	1	-0.4638	0.4300	-1.3066	0.3789	1.16	0.2807
male	1	-0.4778	0.2337	-0.9360	-0.0197	4.18	0.0409
educ	1	0.0440	0.0336	-0.0220	0.1099	1.71	0.1911
lpcappinc	1	0.0462	0.1466	-0.2412	0.3335	0.10	0.7528
chronic	1	0.2522	0.2536	-0.2448	0.7493	0.99	0.3199
canc	1	0.6100	0.4735	-0.3180	1.5381	1.66	0.1976

We also attempted the same regressions for willingness to pay for the 1 in 1000 risk reduction. Table D.11 reports our broadest specification for this model, which includes the age dummies, the gender dummy, the log of income per household member, and the CHRONIC and CANCER dummies. The coefficients on the age dummies suggest a quadratic relationship with log WTP, but this association is not statistically significant. We reach this conclusion by (i) looking at the t statistics on the individual coefficients on the age dummies, and (ii) doing a likelihood ratio test of the null that these coefficients are jointly equal to zero.

The only significant coefficient is that on the gender dummy. The estimated coefficient implies that, holding all else the same, males are willing to pay about 38% less than females for a risk reduction of this size. Income is no longer significant (and its coefficient is very small), and education has a modest positive association with willingness to pay.

VIII. Pooling the data

To increase the number of observations and hence the efficiency of the estimates, we pooled the WTP data. In other words, each respondent contributes two interval-data observations to the sample, one for the 5 in 1000 risk reduction and one for the 1 in 1000 risk reduction. We then ran the same specifications of the interval-data model as in Table D.10, making sure to include a dummy (LARGEREDUCT) taking on a value of one if WTP refers to the 5 in 1000 risk reduction. We expect the coefficient of this dummy to be positive and significant.

Table D.12 shows that indeed the coefficient on this variable is positive and significant at the 1% level. Its magnitude implies that, all else the same, WTP for the 5 in 1000 risk

[15] This criterion results in 281 respondents, 8 of whom do not overlap with the sample created by excluding observations with FLAG1=1.

reduction is about 1.3 times greater than that for the 1 in 1000 risk reduction. Column (A) of Table D.12 shows that the relationship between log WTP and age is quadratic and has an inverted-U shape. This effect is significant and robust to adding other regressors, although in specifications (B) and (C) the t statistics of the coefficient on the age variables are smaller. Column (B) and (C) show that education and income are not important predictors of WTP, but the presence of cancer raises WTP by about 85 percent relative to a person without such ailment.

Table D.12. WTP for an immediate risk reduction. Cleaned sample. T statistics in parentheses.

	(A) Effect of age N=562	(B) Age and individual characteristics N=540	(C) Age, individual characteristics and health status N=540
intercept	6.4283*** (37.49)	5.8272*** (15.17)	5.7325*** (14.75)
5 in 1000 risk reduction (dummy)	0.8342*** (4.89)	0.8526*** (4.86)	0.8520*** (4.85)
Age 50 to 59	-0.3715 (-1.76)	-0.3844* (-1.75)	-0.4482** (-2.00)
Age 60 to 69	0.0780 (0.34)	0.1184 (0.50)	0.0903 (0.37)
Age 70 and older	-0.6986*** (-2.73)	-0.5058* (-1.77)	-0.6017** (-1.96)
Male		-0.4481*** (-2.64)	-0.4262*** (-2.50)
EDUC		0.0355 (1.44)	0.0370 (1.50)
Log income per household member		0.1359 (1.25)	0.1354 (1.25)
CHRONIC			0.1243 (0.68)
Cancer			0.6227* (1.76)
Log likelih.	-593.11	-567.11	-565.09

* = significant at the 10% level. ** = significant at the 5% level. *** = significant at the 1% level.

IX. Future risk reduction

A Weibull interval-data model without covariates pegs mean WTP for a risk reduction of 5 in 1000 at age 70 at 557.63 EUR (standard error of the estimate 175.27). Median WTP for the same risk reduction is 225.95 EUR (standard error 37.87).

Since we have relatively few observations on WTP for latent risk (N=187),[16] we report relatively simple specifications of the WTP regression model in Table D.13. Column (A) of the table suggests that age has little effect on WTP, and that the only significant regressor is the gender dummy. Even income is not significant in this run. Column (B) shows that WTP does depend on the age to which the respondent expects to live, and specification (C) attempts to relate WTP with the subjective probability of surviving to age 70 reported by the respondent during the survey. No significant association is found in the latter case.

Table D.13. WTP Future risk reduction. Interval-data model. Cleaned sample. N=187. T statistics in parentheses.

	(A)	(B)	(C)
Intercept	6.0720*** (4.57)	3.4651** (2.25)	5.2957*** (12.56)
Age	-0.0034 (-0.14)		
Age to which respondent expects to live		0.0285* (1.74)	
Male	-0.8000*** (-2.76)	-0.6920** (-2.36)	
Log income per household member	0.1017 (0.32)	0.0711 (0.41)	
Education	0.0105 (0.025)	0.0185 (0.45)	
Subjective chance to survive to age 70			0.0089 (1.54)

* = significant at the 10% level. ** = significant at the 5% level. *** = significant at the 1% level.

X. Conclusions

The Italy data satisfies the scope test, indicates that WTP depends on gender and income, and—at least when the WTP responses for the 5 in 1000 and 1 in 1000 risk reductions are pooled to increase the sample size—suggests that the relationship between age and log WTP is an inverted-U.

[16] This question was asked only of respondents of ages 60 and younger.

Table D.14. Double-bounded regressions of WTP for the 5 in 1000 risk reduction. City dummies included. Omitted city is NAPOLI (Naples). N=270.

Log Likelihood -264.9692456

Analysis of Parameter Estimates

Variable	DF	Estimate	Standard Error	Chi-Square	Pr > ChiSq	Label
Intercept	1	6.06348***	0.60668	99.8899	<.0001	Intercept
torino	1	1.27198**	0.59699	4.5396	0.0331	
genova	1	2.06389***	0.66687	9.5783	0.0020	
Venezia	1	1.22666**	0.61603	3.9650	0.0465	
milano	1	0.72520*	0.40539	3.2001	0.0736	
age5059	1	-0.29738	0.33654	0.7808	0.3769	
age6069	1	-0.30489	0.36153	0.7112	0.3991	
ag70plus	1	-0.89359**	0.42691	4.3813	0.0363	
male	1	-0.38196	0.25582	2.2293	0.1354	
educ	1	-0.0045509	0.03795	0.0144	0.9045	
lpcappinc	1	0.28959*	0.17418	2.7641	0.0964	

* = significant at the 10% level. ** = significant at the 5% level; *** = significant at the 1% level. Age5059=dummy equal to one if the respondent is of ages 50 to 59; age6069=dummy equal to one if the respondent is of ages 60 to 69; ag70plus=dummy equal to one if the respondent is 65 years old or older; educ=years of schooling; pcappinc=household income in thousand euros divided by number of household members.

Table D.15. Double-bounded regressions of WTP for the 5 in 1000 risk reduction. City dummies included. Omitted city is NAPOLI (Naples). N=270.

Log Likelihood -264.8033587

Analysis of Parameter Estimates

Variable	DF	Estimate	Standard Error	Chi-Square	Pr > ChiSq	Label
Intercept	1	6.05399***	0.63707	90.3041	<.0001	Intercept
torino	1	1.24254**	0.60185	4.2623	0.0390	
genova	1	2.01097***	0.66935	9.0261	0.0027	
Venezia	1	1.18965**	0.61951	3.6876	0.0548	
milano	1	0.71843*	0.41521	2.9938	0.0836	
age5059	1	-0.30297	0.34330	0.7788	0.3775	
age6069	1	-0.28003	0.37152	0.5681	0.4510	
ag70plus	1	-0.87806*	0.45537	3.7181	0.0538	
male	1	-0.36597	0.25903	1.9961	0.1577	
educ	1	-0.0019990	0.03869	0.0027	0.9588	
lpcappinc	1	0.28004*	0.17448	2.5760	0.1085	
chronic	1	-0.04478	0.27614	0.0263	0.8712	
canc	1	0.28825	0.54941	0.2753	0.5998	

*= significant at the 10% level; ** = significant at the 5% level; ***=significant at the 1% level. Age5059=dummy equal to one if the respondent is of ages 50 to 59; age6069=dummy equal to one if the respondent is of ages 60 to 69; ag70plus=dummy equal to one if the respondent is 65 years old or older; educ=years of schooling; pcappinc=household income in thousand euros divided by number of household members.

Appendix 3 The Survey Instrument

The survey instrument adopted by the research team is computer-based. In order to give a better impression of the instrument to the reader, below we present a number of screens that are key to the survey. These screens are taken from the US version of the survey, thereby explaining some of the phrasing etc. The content of the survey instrument components are given in Section 2 above and we adopt this structure when presenting the key screens below.

Component 1 Introduction to the survey, and reassurance that it is not a marketing exercise but that the respondents' opinions are being sought. Questions relate to the respondent's age and gender.

Component 2 Establishment of health status, in which the health of relatives and the individual are recorded, focusing on the presence or absence of various chronic diseases. This has several purposes. The questions are straightforward and therefore help to get the respondent used to the screens; they encourage the respondent to think about their health *before* responding to the WTP questions. The respondent is asked how they think their health status will be in 10 years and when aged 75, relative to their current health. They are asked whether mother and father are still alive, and their ages (if alive). They are also asked to what age they think they will live.

Component 3 This component educates the respondent about probabilities in general and specifically about risks of death. The main purpose of this section is to communicate facts about probabilities clearly and test for comprehension, eschewing tests of mathematical ability. Screens move from simple coin flips to a roll of the die ***(Screen 14 below)*** and then introduce the idea of a grid, the total number of squares representing possible outcomes, and red squares representing outcomes of a particular type. A key graphic – 1,000 grid squares, with several coloured red, represents the risk of death ***(Screen 17 below).***

Understanding of the concept of risk is tested by first describing two people, Person 1 and Person 2. These people are identical in every way, except one has a 5 in 1,000 chance of dying over the next 10 years while person 2 has a 10 in 1000 chance of dying over the next 10 years. The respondent is shown side-by-side graphs of the risks for these people and asked to pick which person has the largest chance of dying. If the respondent answers this question incorrectly (s)he is asked a question identical except that different probabilities are used ***(Screen 18b below)***. Even if a respondent can distinguish these risks, he or she may not feel that the difference in risk is "significant." To identify such respondents, it is asked which of these two people they would rather be (including "indifferent" as a possible answer).

Component 4 This component provides baseline risks, using the respondent's age and gender information, and additional information about these risks to put them into context. The idea of baseline risks is introduced by showing the effect of age on baseline risks in ten-year increments, both verbally and with a graph. The respondent sees a grid with the appropriate number of red squares representing the 10-year baseline risks for someone of their age and gender ***(Screen 22a below).*** To help fix this baseline in the respondent's mind, he or she is asked to create his or her own baseline risk graph by pushing a key.

Component 5. Information is presented to the respondents on age- and gender-specific leading causes of death and common risk-mitigating behaviour – both medical and non-medical. Illustrative risk reductions for these are provided (estimated from the literature) along with cost ratings ***(Screen 31m below).*** The idea is also introduced that even though a procedure or action may be free to the insured, someone still pays.

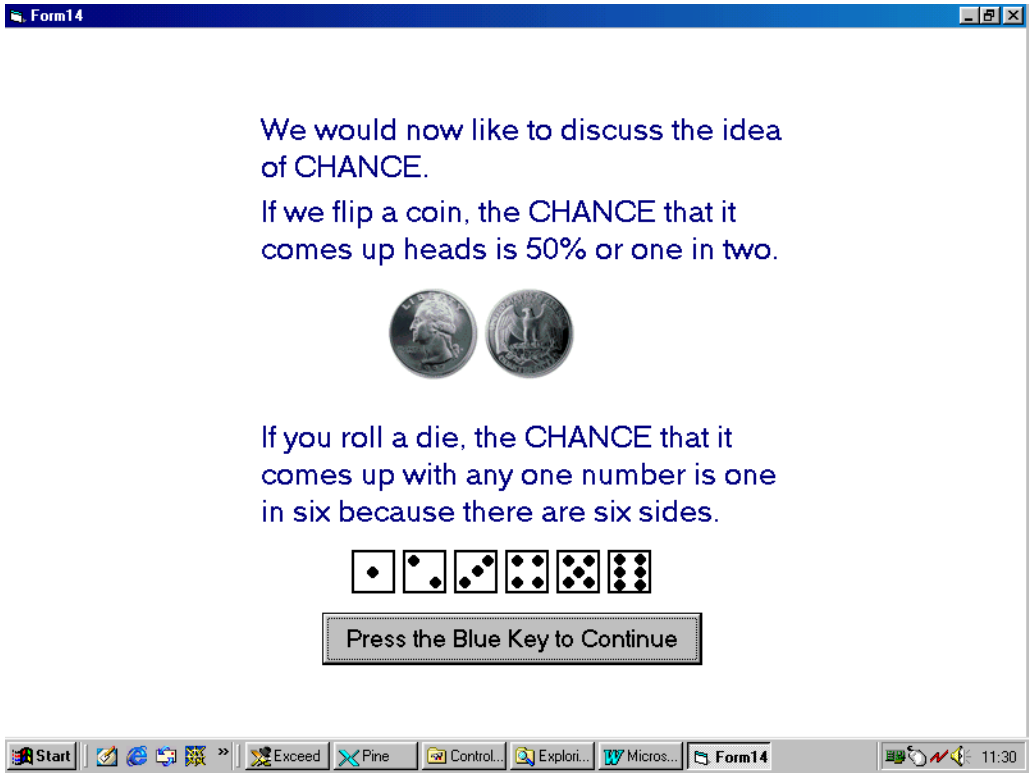
Component 6. This component seeks to elicit WTP for risk reductions of a given magnitude, occurring at a specified time, using dichotomous choice methods with one follow-up.

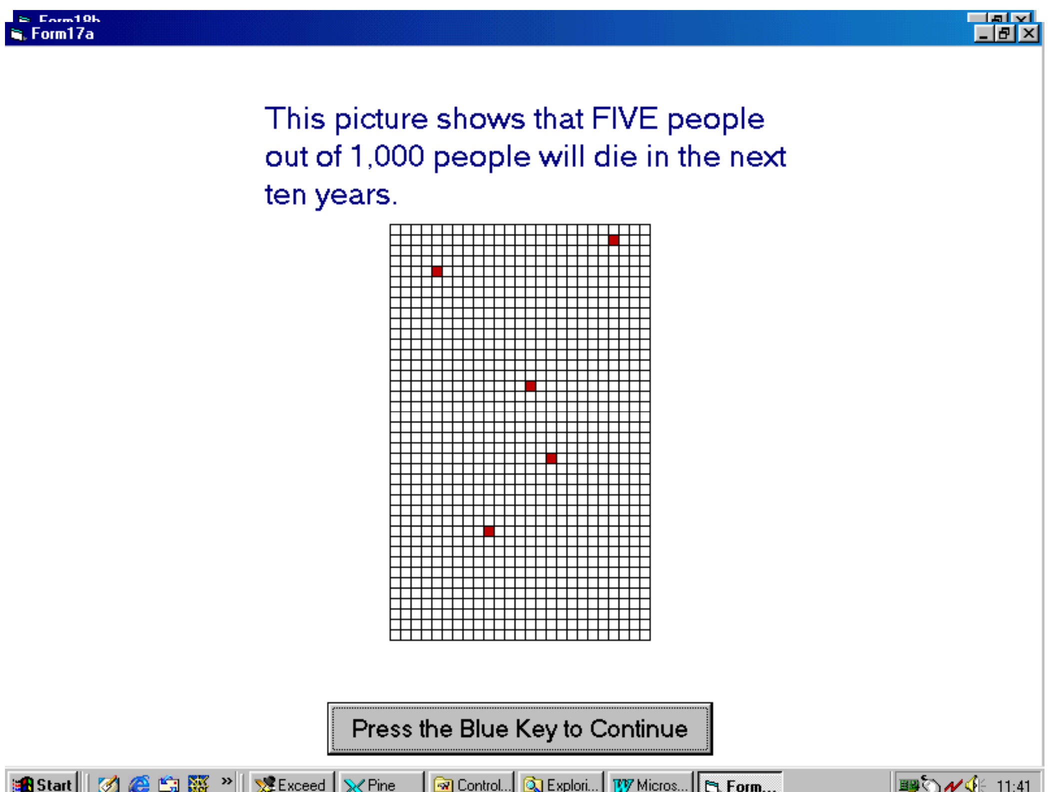
WTP per year over the next 10-year period is asked for a risk change of 5/1,000 over the same period, and 1 in 1,000 over the same 10-year period. The 10-year sum of the annual payments is also provided. For the 5/1,000 risk change these questions are shown in ***Screens 33 and 37 below***, where the question is expressed in text and the payment period and period over which the risk reduction occurs is shown pictorially. For the third WTP question (asked only of individuals 60 or less), the respondent is then told his or her gender-specific chance of dying between ages 70 and 80 and is asked, through dichotomous choice questions, their WTP each year over the next ten years for a future risk reduction beginning at age 70 and ending at age 80 *which totals* 5 in 1,000. The respondent is reminded that there is a chance he or she may not survive to age 70, making a payment today useless. ***Screens 49, 51 and 53 below*** show how this question is presented to the respondent. He or she is then given the opportunity to revise their bid. During an extensive debriefing section of the survey, the respondent is asked whether they thought about their health state during this future period. Each WTP question is followed by a screen, (an example of which is ***Screen 48 below***), to gauge the strength of a respondent's conviction in his WTP responses.

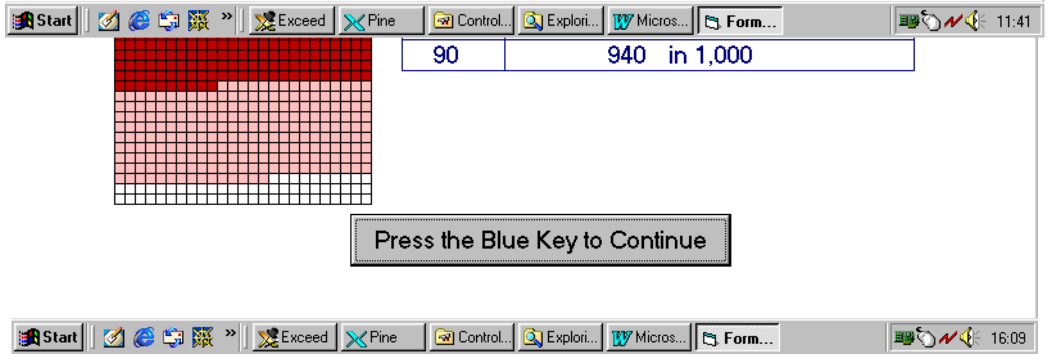
Component 7. This includes debriefing questions. Each debriefing question probes the state of the respondent's mind when they answered the various WTP questions and some other questions. These included:

- understanding of idea of 'chance' (***Screen 58 below***)
- acceptance of specific baseline?
- specific product in mind? If yes – what kind of product?
- Doubts about product? If yes – influence WTP?
- Did you think you would suffer any side-effects?
- Did you consider whether you could afford payments?
- Did you think of other benefits? (***Screen 66 below***) If yes - to yourself, others, for you living longer or improved health. If yes – influence WTP? – raise/lower?
- On WTP 70-80 did you consider whether – would live to age 70? Or your health at age 70?
- Household Income

The short form SF36 questionnaire used on a standard basis in personal health assessment research was also included in the survey in order to provide more detailed data on health status. Finally, the respondent is given the opportunity to review the values (s)he has chosen and amend, if (s)he so wishes (***Screen 106 below***).


Form14
We would now like to discuss the idea of CHANCE.
If we flip a coin, the CHANCE that it comes up heads is 50% or one in two.
If you roll a die, the CHANCE that it comes up with any one number is one in six because there are six sides.
Press the Blue Key to Continue
Start
Exceed
Pine
Control...
Explori...
Micros...
Form14
11:30


Form17a
This picture shows that FIVE people out of 1,000 people will die in the next ten years.
Press the Blue Key to Continue
Start
Exceed
Pine
Control...
Explori...
Micros...
Form...
11:41


90
940 in 1,000
Press the Blue Key to Continue
Start
Exceed
Pine
Control...
Explori...
Micros...
Form...
16:09

Form31m

The costs of actions people take to reduce their chance of dying vary by action. YOUR costs also depend on the insurance coverage you have. Even if an action is FREE to you, someone must pay for it. The table below rates some of the actions that we have just mentioned by their annual costs to treat a person, no matter who pays.

Treatment	**Cost Rating**
Controlling Cholesterol Through Drug Treatment	Expensive
Colo-Rectal Cancer Screening	Somewhat Expensive
Controlling High Blood Pressure through Drug Treatment	Expensive
Prostate Cancer Screening	Inexpensive

Press the Blue Key to Continue

Form33

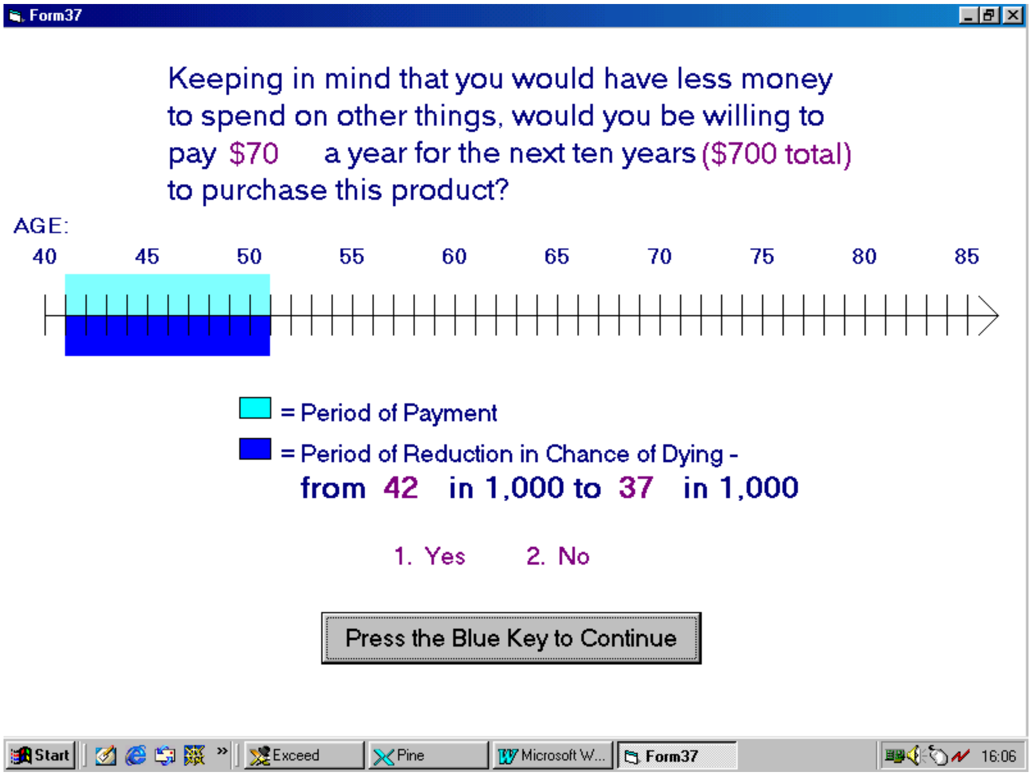
Suppose that a new product becomes available that, when used over the next ten years, would reduce your chance of dying from a disease or illness. This product would reduce your TOTAL chance of dying in the next ten years from:

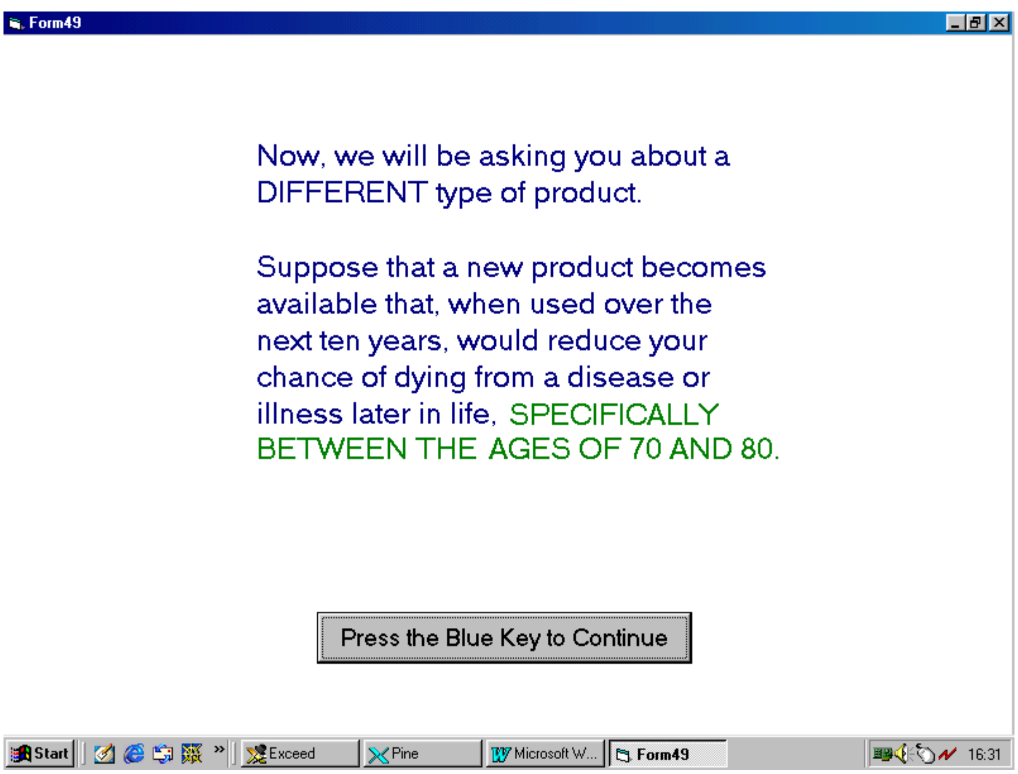
42 in 1,000

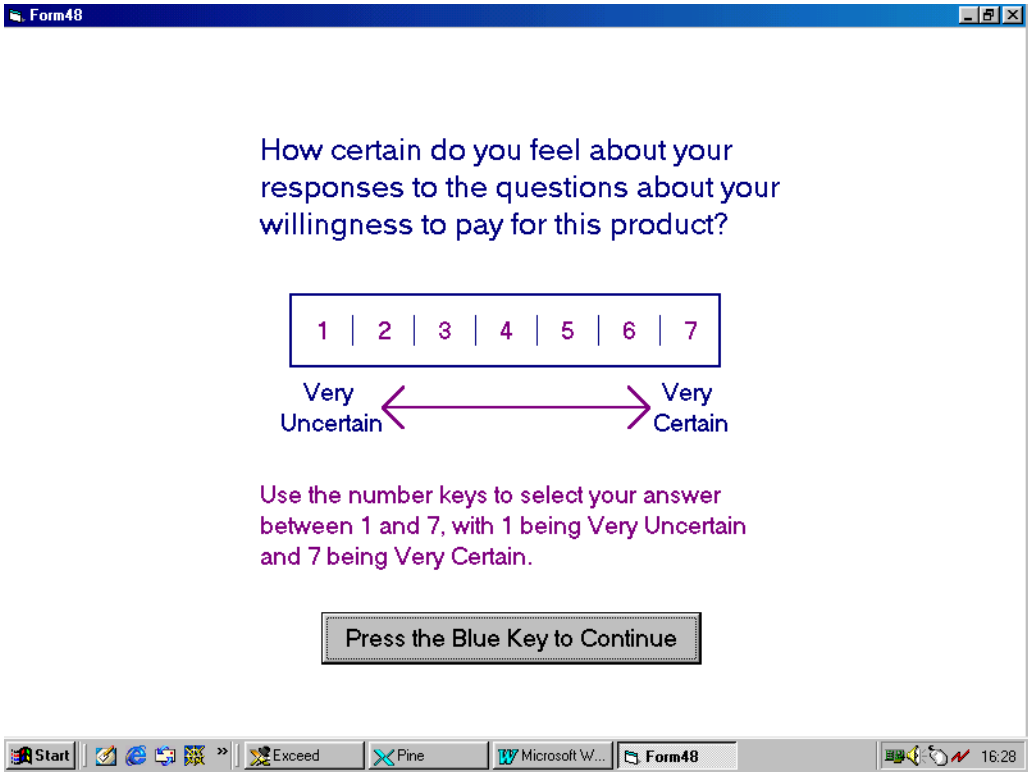
to

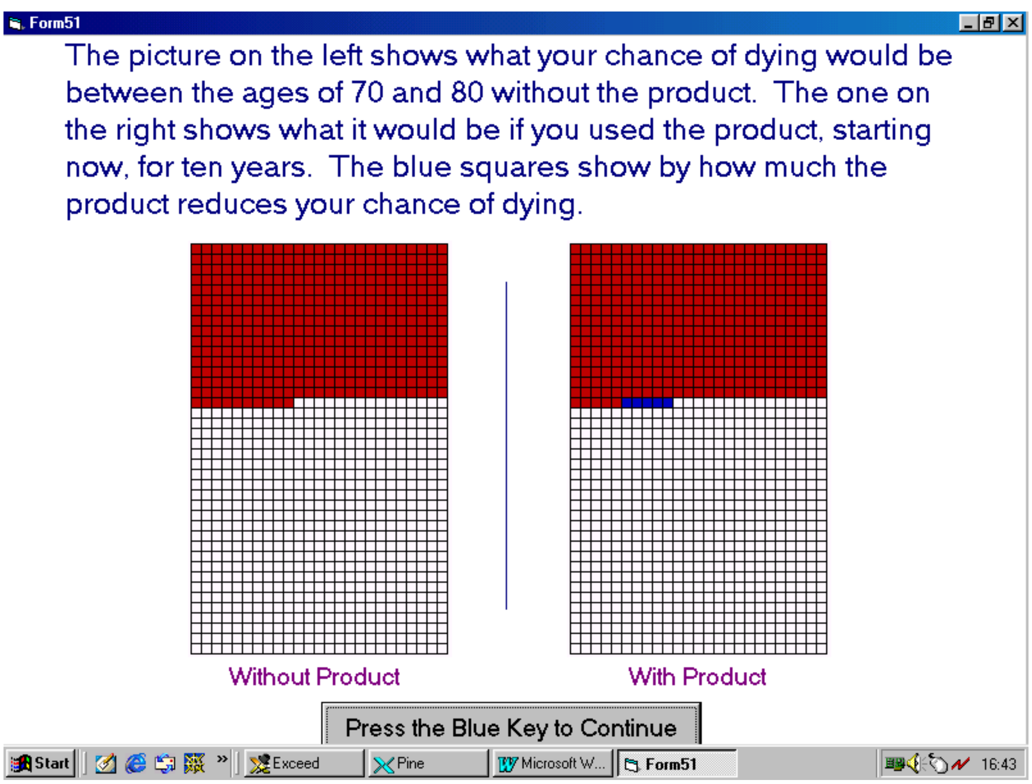
37 in 1,000

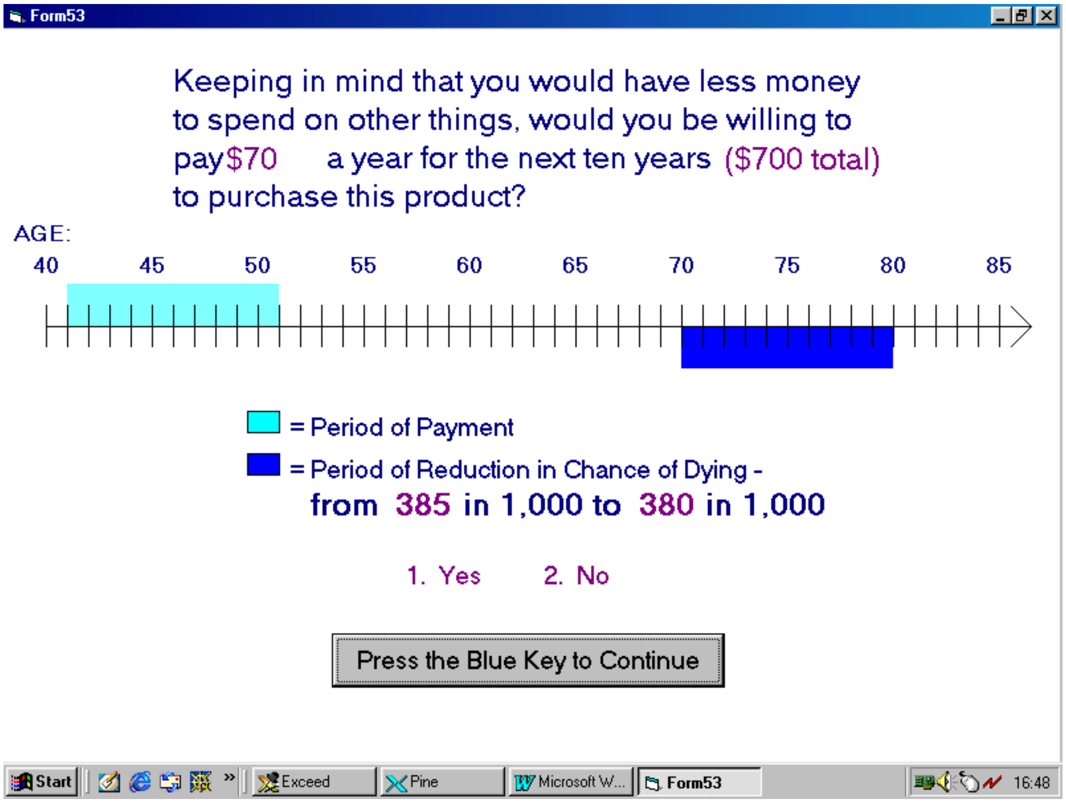
Press the Blue Key to Continue

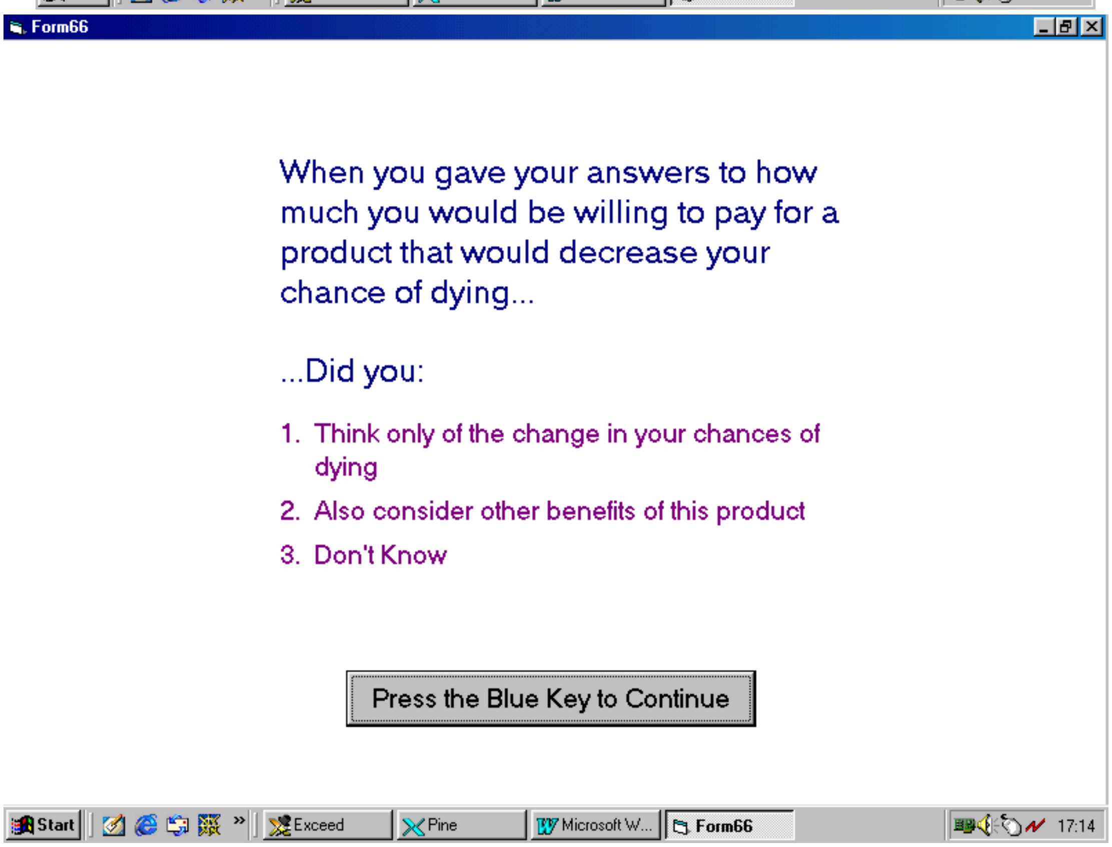

Form37
Keeping in mind that you would have less money to spend on other things, would you be willing to pay $70 a year for the next ten years ($700 total) to purchase this product?
AGE:
40 45 50 55 60 65 70 75 80 85
= Period of Payment
= Period of Reduction in Chance of Dying -
from 42 in 1,000 to 37 in 1,000
1. Yes 2. No
Press the Blue Key to Continue
Start
Exceed
Pine
Microsoft W...
Form37
16:06

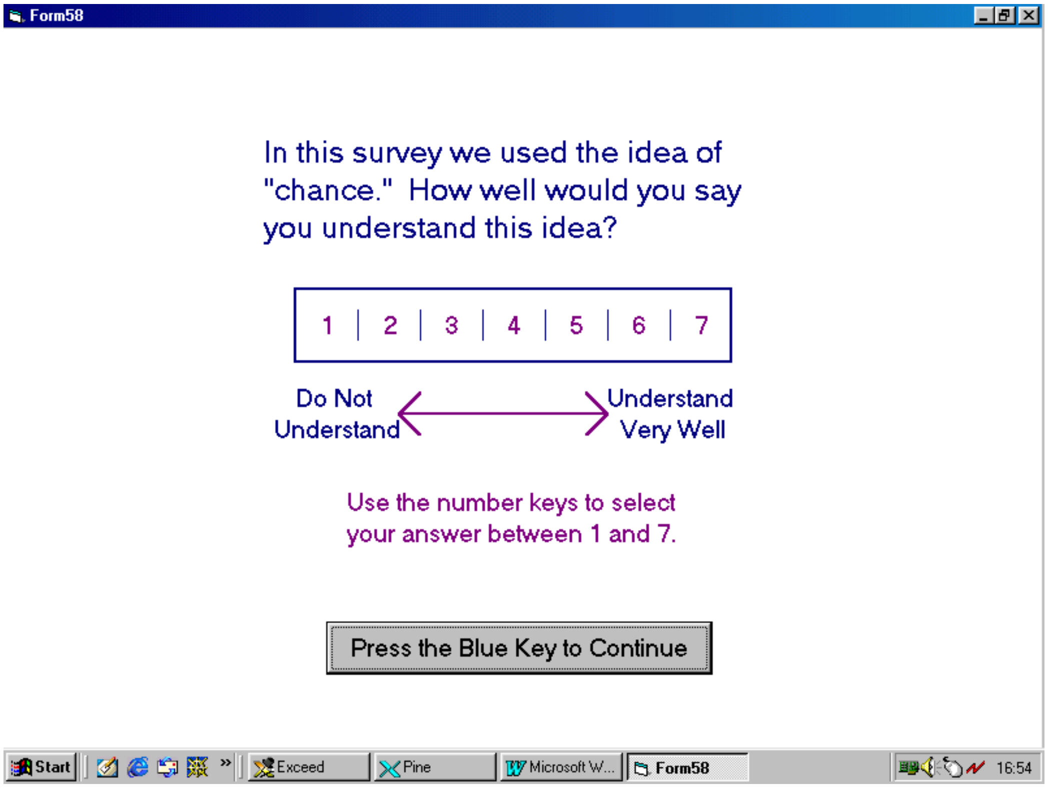

Form49
Now, we will be asking you about a DIFFERENT type of product.
Suppose that a new product becomes available that, when used over the next ten years, would reduce your chance of dying from a disease or illness later in life, SPECIFICALLY BETWEEN THE AGES OF 70 AND 80.
Press the Blue Key to Continue
Start
Exceed
Pine
Microsoft W...
Form49
16:31


Form48
How certain do you feel about your responses to the questions about your willingness to pay for this product?
1 | 2 | 3 | 4 | 5 | 6 | 7
Very Uncertain
Very Certain
Use the number keys to select your answer between 1 and 7, with 1 being Very Uncertain and 7 being Very Certain.
Press the Blue Key to Continue
Start
Exceed
Pine
Microsoft W...
Form48
16:28


Form51
The picture on the left shows what your chance of dying would be between the ages of 70 and 80 without the product. The one on the right shows what it would be if you used the product, starting now, for ten years. The blue squares show by how much the product reduces your chance of dying.
Without Product
With Product
Press the Blue Key to Continue
Start
Exceed
Pine
Microsoft W...
Form51
16:43


Form53
Keeping in mind that you would have less money to spend on other things, would you be willing to pay $70 a year for the next ten years ($700 total) to purchase this product?
AGE:
40 45 50 55 60 65 70 75 80 85
= Period of Payment
= Period of Reduction in Chance of Dying - from 385 in 1,000 to 380 in 1,000
1. Yes 2. No
Press the Blue Key to Continue
Start
Exceed
Pine
Microsoft W...
Form53
16:48


Form66
When you gave your answers to how much you would be willing to pay for a product that would decrease your chance of dying...
...Did you:
1. Think only of the change in your chances of dying
2. Also consider other benefits of this product
3. Don't Know
Press the Blue Key to Continue
Start
Exceed
Pine
Microsoft W...
Form66
17:14


Form58
In this survey we used the idea of "chance." How well would you say you understand this idea?
1 | 2 | 3 | 4 | 5 | 6 | 7
Do Not Understand
Understand Very Well
Use the number keys to select your answer between 1 and 7.
Press the Blue Key to Continue
Start
Exceed
Pine
Microsoft W...
Form58
16:54

IV VALUATION OF ENVIRONMENTAL IMPACTS BASED ON PREFERENCES REVEALED IN POLITICAL NEGOTIATIONS AND PUBLIC REFERENDA

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PSI, Paul Scherrer Institut, 2004Vito, Villingen, Switzerland

Abstract

By analyzing the decisions of policy makers and in addition public referenda, shadow prices for global warming (ca. 5 to 22 € per ton of CO_2) and exceedance of critical loads for eutrophication and acidification (ca. 100 € per hectare of exceeded area and year with a range of 60 – 350 €/ha year) have been developed.
These data are complementary to the valuation steps in ExternE, and can be used to compare energy technologies and fuels, but cannot be used to inform policy makers on the impacts of these environmental problems.

IV.1) Introduction

This chapter relates to the evaluation of environmental impacts based on preferences revealed in :
(1) political negotiations, with an application to
- acidification and eutrophication) and
- global warming,

(2) public referenda, application to global warming

The overall methodology is already explained in the methodology chapter. A general overview of methods and how they relate is given in Table IV-1. The issues dealt with in this chapter are shaded in light grey.

Table IV-1 : Overview of methods used in ExternE and NewExt Work Package 3 to quantify and value impacts.

	Air pollution			Global warming
	pubic health	agriculture, materials	ecosystems	
ExternE , Impact pathway approach				
Quantification of impacts	**YES**	**YES**	**Yes critical loads**	**Yes, partial**
Valuation	**Willingness to pay (WTP)**	**market prices**	**no valuation**	**Yes, WTP & market prices**
NewExt WP 3				
Quantification of impacts			**Yes critical loads**	/
Valuation based on preferences revealed in				
political negotiations			**UN-ECE; NEC**	**Implementin g Kyoto, EU**
public referenda				**Swiss Referenda**

IV.2) Valuation of environmental impacts of acidification and eutrophication based on implicit values of policy makers [1]

IV.2.1) Introduction

The effects of SO_2, NO_X and NH_3 on human health, crops and building materials have been quantified as external costs in the ExternE project. Up to now, ExternE failed however to provide external cost estimates for impacts on ecosystems due to acidification and eutrophication. Acidification is mainly caused by emissions of sulphur dioxide (SO_2), nitrogen oxides (NO_X) and ammonia (NH_3), while eutrophication by airborne pollutants is mainly caused by NO_X and NH_3.

Evidence has shown however that acidification has a potential negative effect on aquatic and terrestrial ecosystems, surface water, agricultural and forestry yields, buildings and human heath. Also eutrophication, or the enrichment by nitrogen nutrients of soil, ground- and surface water, results in a potential negative effect on aquatic and terrestrial ecosystems, surface water and agricultural and forestry yields.

The external costs accounting framework does not properly address the environmental impact categories which are the main driving force for some of the most important international energy and environmental policy actions (EU acidification strategy, EU NEC directive, UN-ECE LRTAP protocols, etc.)

This study explores the possibilities to value the impacts of airborne emission of SO_2, NO_X and NH_3 on acidification and eutrophication of ecosystems using the implicit values of policy makers, also called the *standard price* approach, i.e. to use the abatement costs of emissions reductions as a proxy for the revealed willingness to pay of European society for the improvements in ecosystems health.

IV.2.2) Methodology

- **The standard price approach, an approach based on the implicit values of policy makers**

The *standard price* approach estimates the revealed preferences of policy makers. It calculates the benefits of emission reduction – as perceived by policy makers - based on the abatement costs (Ct_1) to reach a well-defined emission reduction target (Et_1) (*cf.* Figure IV-1). These costs are a proxy for the benefits that policy makers attribute to these reductions, as we assume that policy makers act as rational decision makers who carefully balance (their perception of) abatement costs of emission reductions with (their perception of) the benefits of these emissions.

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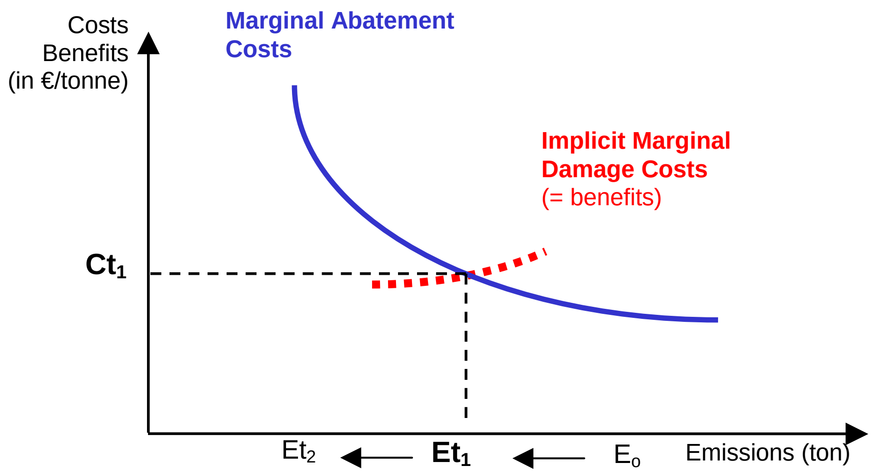
1 This part is based on: De Nocker, Leo; Vermoote**,** Stijn: Valuation of environmental impacts based on preferences revealed in political negotiations: Applications to impacts of air pollution to ecosytems and global warming, Vito, 2004.



Legend :

E_0 : Initial emission level that needs to be reduced

Et_1 : Negotiated emission reduction target for a certain emission reduction program

Et_2 : Emission reduction target that was not withheld during the negotiations

Ct_1 : Marginal abatement costs associated with the negotiated emission reduction target Et_1 and thus corresponding to the implicit benefits as perceived by the policy makers (~ the willingness-to-pay to reach the goals set by the emission reduction program associated with Et_1)

Figure IV-1 : Principle of the *standard price* method

It has to be noted that this approach is somewhat different from the valuation step used so far in ExternE, as the latter mainly reflects WTP of individuals, measured by a wide number of indicators and methods. These methods may include the estimation of “revealed” preferences of citizens, e.g. by using data of additional costs of safety equipment (e.g. in cars) to evaluate the WTP of people for reducing risks.

The key difference between the *standard price* approach and ExternE is that the first estimates the preferences of policy makers, as an indicator of the preferences in society. Ideally, we would wish to estimate the preferences and WTP of the individuals for (marginal) improvements of ecosystems health in Europe, e.g. by means of CVM techniques. As the latter is not possible for this study, we use implicit values of policy makers, as a *second-best* method.

So far, these implicit values of policy makers have not been used within ExternE, although the ‘shadow prices’ that administrations use for the reduction of risks in different policy areas (transport, public health) have been used to ‘validate’ the WTP for reduction of mortality risks based on studies that evaluate the WTP of individuals (Bickel *et al*., 2000).

As the *standard price* approach is based on the current preferences of policy makers, as reflected in air quality policies, it cannot be used for cost-benefit analysis or policy advices related to these emission reduction policies. Nevertheless, this *second-best* method gives useful data for comparison of energy technology and fuels because it gives us ‘shadow prices’ for a non-market scarcity, i.e. protected ecosystems from acidification and eutrophication.

IV.2.2.1 Estimating the shadow prices per ton pollutant for impacts on ecosystems

Earlier studies have used abatement costs as 'shadow prices' for the total impacts on human health, agriculture and ecosystems, expressed as € per ton pollutant. We follow a more sophisticated approach, which aims at figures that are more in line with the *impact pathway* approach of ExternE and that are additive to the ExternE estimates for impacts on human health, agriculture and building materials. Therefore, the analysis combines the impact pathway approach to estimate impacts in physical terms (step 1), which are then valued following a careful analysis of international agreements of emission reductions in Europe (step 2). On this basis, we can estimate the shadow price per ton of emissions (step 3).

Figure IV-2 shows the different steps.

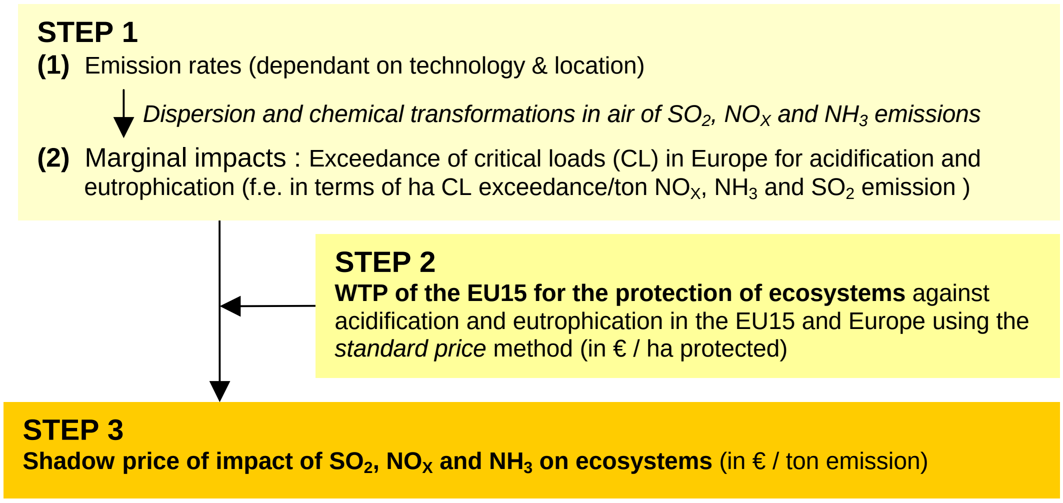


Figure IV-2: Steps in the analysis to determine the shadow price for impacts of acidification and eutrophication on ecosystems.

IV.2.2.2 Selection of the emission reduction programs and determination of the WTP

The implicit values of policy makers associated with the protection of ecosystems have been defined in terms of pushing back (closing the gap) of the number of hectares of ecosystem that remain unprotected to acidification and eutrophication. Therefore, in step 2 of the analysis, we have to determine the society's WTP for one hectare of ecosystem protected. The stepwise application of step 2 is presented in Figure IV-3.

The calculations of step 2 result in one figure for the whole of EU-15 for each examined emission reduction program.

The different assumptions made and parameters used in the analysis are explained in the next paragraph.

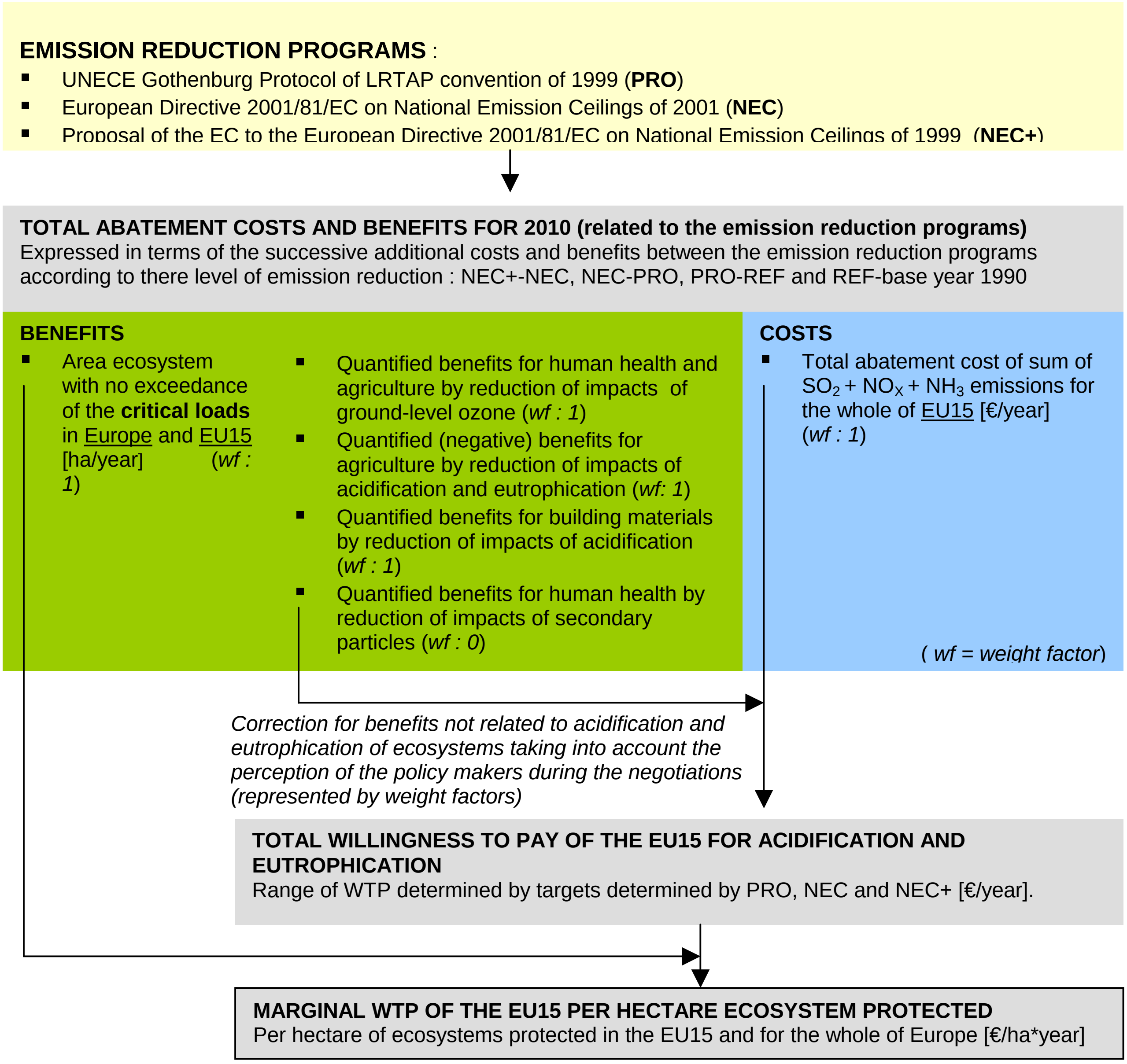


Figure IV-3 : Detail of the stepwise application of step 2 of the analysis

As Figure IV-3 indicates, we have to define a marginal cost curve for emission reductions, and select an emission reduction level which has been agreed upon by the policy makers.

To this purpose we have analysed two emission reduction programs and a reference scenario for the year 2010:

- **Reference scenario** (**REF**): The 'reference' scenario can be seen as a target for 2010 based on a 'business as usual' scenario starting for the status in 1998. (Amann *et al*., 1999a)
- **Protocol of Gothenburg on the Convention on Long-range Transboundary Air Pollution (1999)** (**PRO**): The policy goals of PRO are based on making significant progress towards reaching a scientific based objective, i.e. a reduction by 50% of the number of hectares of ecosystems facing an exceeding of their 'critical loads' for eutrophication and acidification for the year 2010. (UN-ECE, 1999)

- **European directive 2001/81/EC on National Emission Ceilings for some air pollutants** (**NEC**): The original proposal on national emission ceilings of the European Commission (**NEC+**) of 9 June 1999 was much more ambitious and scientifically better grounded than the PRO. The directive finally adopted in 2001 (NEC) was set less ambitious so that the emission levels are only slightly stricter for NEC than for PRO. NEC has the same policy targets as PRO. (European Commission, 2001)

PRO and NEC are both based on a 'multi-source, multi-effect approach', taking into account a multitude of sources and locations of emissions and a multitude of receptors and locations for deposition. The policy does not only focus on the effects of acidification and eutrophication by SO_2, NO_X and NH_3 on ecosystems, but also those of ground-level ozone by NO_X and VOC emissions on human health, agriculture and ecosystems.

IV.2.2.3 Basic assumptions made in the analysis

The following basic assumptions have been taken into consideration in the valuation of environmental impacts of acidification and eutrophication on ecosystems:

The major parameters and assumptions related to ecological indicators

- The number of hectares of ecosystem, for which critical loads for acidification and eutrophication have been exceeded, has been used as the physical indicator to valuate the effects of acidification and eutrophication on ecosystems. This study does not question the use of the critical loads approach as a physical indicator. Although this is in line with the indicators used in a wide range of scientific and policy documents, it does not fully reflect all marginal impacts on all ecosystems.
- We simply add up exceedance of different types of ecosystems, both terrestrial and aquatic, and we add up impacts of acidification and eutrophication.
- The number of hectares of ecosystem for which the critical loads are exceeded are evaluated for the whole of the EU15, non-EU and Europe. Hereby, regional differences in critical loads and the extent in which the critical loads are exceeded are not accounted for.
- We use a single value for all ecosystems, irrespective of its characteristics and location. This simplification is characteristic for the valuation based on the implicit values of policy makers on EU level.

The major parameters and assumptions related to costs indicators

- We assume that the costs as estimated by the technical-economic models are a good indicator for the WTP (*cf.* Amann *et al.*, 199a and 1999b; Holland *et al.* 1999a and 1999b). Although it is an important element, policy makers also take other cost issues into account, including the impact of the measures on economy, employment, distribution of incomes, etc.
- We do not use marginal costs of single measures but the average costs of a marginal policy package. Although these values are lower than the marginal costs of the individual measures, it better reflects the package deal in decision making and its results are less sensitive to small changes in emission reduction scenarios or estimates of costs for single measures. Although it is true that marginal costs of additional measures are much higher, we cannot consider these higher costs to reflect the real WTP of politicians, as they were not willing to accept policy packages like the initial NEC proposal (NEC+) with much higher marginal costs per ton or hectare (*cf.* Figure IV-4).

The major parameters and assumptions related to other impact categories

- The range of WTP values is determined by weight factors (0 or 1), representing the perception of policy makers on the importance of a certain effect during the negotiations on PRO and NEC.
- We have assumed that the ExternE-based[2] estimates for the effects of ozone are fully believed by policy makers and have been taken into account (*cf.* Table IV-1). The WTP is corrected for all of the impacts of ozone because the emission reduction programs clearly define targets for AOT40 and AOT60 (*weightfactor = 1*).
- Setting targets for critical loads for acidification and eutrophication and for ozone (AOT40) also affects crop yields. Defining targets for SO_2 emission curbs is also beneficial for the protection of building materials. We have corrected the WTP for benefits for agriculture and building materials (*weightfactor = 1*). This assumption is not that important as benefits for agriculture and building materials are relatively small.
- Although the studies indicate that there are big potential benefits from the emission reduction programs on health impacts from secondary particles (aerosols), we have not used these data to correct the abatement costs for this benefit (*weightfactor = 0*).
    - A first reason is that it was not the objective of the agreements to tackle the issue of ambient particles (*cf.* Table IV-1). The major goal of the Gothenburg protocol and the NEC directive for 2010 is, next to abatement of ground-level O_3, ecosystem protection, i.e. a 50% gap-closure of the accumulated exceedance of the critical loads for acidification and eutrophication. Although both emission reduction programs mention the ‘additional’ benefit of a reduction of the formation of secondary particulate matter (aerosols) by SO_2 and NO_X emission curbs, this benefit did very likely not play a major role in the definition of the emission reduction targets for SO_2 and NO_X. This conclusion is based on the analysis of the official text of the Gothenburg protocol and the legal text of the NEC directive. This assumption is also checked by the execution of a questionnaire with a small selection of key players that have been involved in the formulation of air pollution. It is not possible to draw strict conclusions of this exercise but for the results obtained so far, we can conclude that secondary particles did play an important role during the negotiations on the Gothenburg Protocol and NEC directive but rather in an ‘implicit, qualitative’ way than in a ‘tangible, quantitative’ way.
    - Second, although one may argue that the secondary particles effect had some impact on the negotiations, it is doubtful that these benefits got the same weighting as the ExternE numbers would suggest. The most important numbers (on chronic mortality) have a high uncertainty rating in cost-benefit analyses executed for the Gothenburg Protocol and the initial proposal on the NEC directive, as indicated in the reports of Holland *et al.* (1999a, 1999b).
    - Third, if public health played a decisive role, and if the numbers were taken into account, policy makers should have decided on tighter emission standards.
- We have taken into account the benefits of ozone on public health and agriculture, but not the impacts of ozone on ecosystems (*weightfactor = 0*) (*cf.* Table IV-2).

[2] Although the data related to the benefits of the emission reduction scenarios are not identical to the ExternE data, there are based on similar methodologies, dose-response functions and valuation principles as the ExternE accounting framework. An important issue however is the presentation of the numbers in classes of uncertainty.

The major generic assumptions and those related to selection of scenarios

- The 'reference' (REF) scenario has not been used to determine the range of the willingness-to-pay for improvement of ecosystems, as it includes measures focusing to other impact categories, and the costs may not be comparable to these of other scenarios.
- We have based the WTP on the UN-ECE Gothenburg protocol and the EU directive on NEC (NEC), as the policy makers have reached an agreement on these emission reduction programs.
- The initial NEC proposal (NEC+) represents an upper margin for the WTP. NEC+ was not agreed upon by the Council but it was well founded and can be seen as a minimal interim goal if the EU long-term targets of no exceedance of the critical loads want to be reached in 2020.

Table IV-2 : Presentation of the impact categories affected by the emission reduction targets and the impact categories taken into account in the determination of the WTP/ ha ecosystem protected in relation with the priority goals of the UN-ECE Gothenburg Protocol (1999) and the EU directive on National Emission Ceilings (2001).

		PRIORITY GOALS OF PRO AND NEC					
Concrete targets for:		**CL acid**	**CL eutro**	**AOT40**	**AOT60**	**Additional benefits of PRO and NEC**	
By defining emission reduction targets for:		NO_X, SO_2	NO_X, NH_3	NO_X, VOC (O_3)	NO_X, VOC (O_3)	SO_2	SO_2, NO_X: PM (indirect)
Impact categories	Ecosystems	✓	✓	✓*			
	Crops	✓	✓	✓			
	Human health				✓	✓	✓**
	Materials					✓	✓

<u>Legend:</u>

CL acid: Critical loads for acidification

CL eutro: Critical loads for eutrophication

AOT40: Accumulated excess exposure over threshold of 40 ppm O_3 (critical level for vegetation protection)

AOT60: Accumulated excess exposure over threshold of 60 ppm O_3 (critical level risk to human health)

PM: particulate matter (aerosols), indirect benefit by emission curbs of its precursors SO_2 and NO_X

✓: Impact categories experiencing benefits by the emission curbs needed to reach a particulate target.

✓: Impact category for which the WTP/hectare ecosystem has been corrected for in the analysis.

*: Impacts on ecosystems due to precursor emissions of ozone are not included in analysis because these impacts have not been quantified.

**: Important potential benefit, but our assumption is that it was not taken into account in the negotiations.

IV.2.3) Results and discussion

IV.2.3.1 WTP per hectare for improvement of ecosystems health.

In Table IV-3 the 'marginal' WTP range is presented and compared with the 'marginal' emission costs of the reference scenario (REF). It is important to note that this WTP range concerns the willingness-to-pay of the EU15 as a whole.

Table IV-3 : The 'marginal' WTP for the EU15 per hectare of ecosystem protected (in EUR/ha*year)

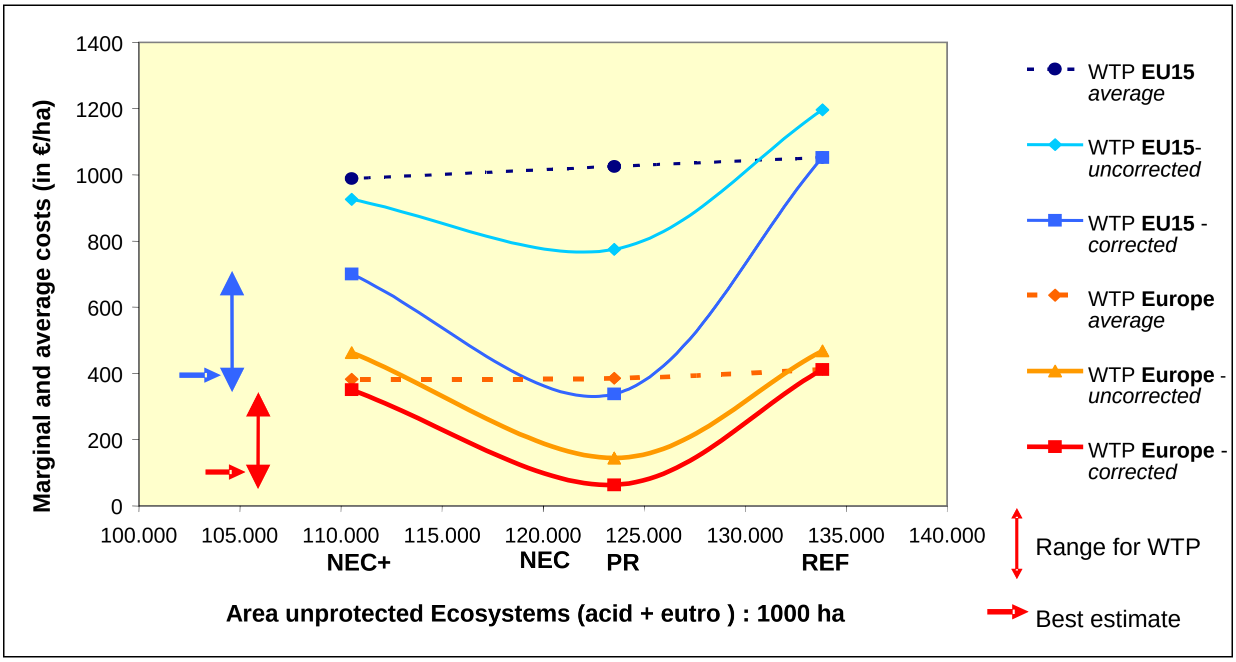
	REF (1)	*proxy WTP/ha* PRO (2)	NEC	*WTP/ha $_{MAX}$* NEC+ (3)
Corrected for other benefits*				
Per ha in Europe	466	**63**	/	**350**
Per ha in EU15	1.052	338	/	674
Uncorrected				
Per ha in Europe	469	145	/	463
Per ha in EU15	1.196	775	/	926

Remarks:

- Corrected for other benefits categories, according to the weight factors and based on abatement costs.
- The table shows the full range. The numbers in bold indicate the range for the best estimate, following the assumptions as discussed in the text.
- Per ha in EU15: analysis based on analysis limited to area ecosystems in the EU15.
- Per ha in Europe: analysis based on impacts for area ecosystems all over Europe.

(1) Based on total costs for the reference scenario, compared to 1990.
(2) Based on total additional cost between PRO and REF.
(3) Based on total additional cost between NEC+ and PRO, calculated by extrapolation from benefits calculated on European scale.

As there are arguments to base the WTP on each of the emission reduction programs (PRO, NEC or NEC+), we report the range of possible values for WTP per hectare (Table IV-3). The numbers in bold indicate the range for the best estimate, ranging from 63 to 350 €/hectares of ecosystems protected in Europe. If we calculate the WTP per hectare only for those ecosystems in the EU15, then these values go up from 338 to 674 €/ha. If we would not correct for the other benefits categories, then all these values would be higher (cf. Table IV-3 and Figure IV-4).



Legend :
- WTP - uncorrected = WTP based solely on the abatement costs to be made by the EU15 member states per ha protected ecosystem
- WTP - corrected = WTP per ha protected ecosystem based on the abatement costs to be made by the EU15 member states and corrected for other benefits (ground level ozone, benefits for agriculture and buildings) for the EU15.
- WTP EU15 / WTP Europe = WTP per ha protected ecosystem within EU15 / Europe
- Average = Average costs for emission reduction scenario compared to the base year 1990 (=e.g. total costs of PRO compared to 1990 /total improvement of ha protected ecosystem of PRO compared to 1990)

Figure IV-4 : The 'marginal' and average cost of policy packages for protection of ecosystems (in €/extra hectare protected from acidification and eutrophication) for emission reduction goals as determined by the REF scenario, the Gothenburg Protocol (PRO), the NEC directive (NEC) and the proposal on the NEC directive (NEC+) for 2010.

The lower range represents the additional costs to reach the Gothenburg protocol, the higher values reflect the costs to meet the emissions reductions of the ambitious plan of the initial NEC proposal by the EC (*cf.* Figure IV-4). As the policy makers have reached an agreement on the Gothenburg protocol and NEC, we take these emission reduction programs as the basis for our best estimate, for which we use a rounded number of **100 €/ha** for the 'marginal' WTP per hectare of ecosystems protected in Europe. We assume that policy makers of the EU have the same WTP for improving ecosystems health all over Europe, including both EU15 and non-EU Europe. As the total number of hectares protected in the whole of Europe is more than twice the number of hectares protected in EU15, this leads to a lower estimate for WTP per hectare protected if the whole of the European area of ecosystems protected is evaluated. It does however not affect the shadow price expressed per ton pollutant. As the next step of the analysis, we multiply the WTP figure with the number of hectares affected by emissions from individual countries.

Analysing the costs curves from NEC to NEC+, the costs to reach the more ambitious emission reductions targets increase, especially if expressed as cost/ton emission reductions (not presented in this report). This increase is less sharp if expressed per hectare protected, as the impact of one ton of emission reduction on the number of hectares protected also increases.

One could argue that our best estimate is on the low end, because more costly measures have been decided upon in the reference scenario for many countries. It is difficult to interpret the costs of the REF scenario in comparison to the other scenarios. It is not clear whether this reflects a particularity of the models and data used, or whether it reflects the fact that the reduction measures put into practices are not based on the cost-optimal solutions (expressed in €/ton emission reduction) but on a wider range of criteria, including e.g. effects on public health (transportation sector) or economic impacts of measures. Nevertheless, this may as well illustrate that some member states have a higher WTP. Our approach cannot take this element into account.

On the other hand, one could as well argue that the 100 € per hectares may be an upper estimate, as we did not correct for public health benefits of secondary particles, that we did not take impacts of ozone on ecosystems into account, and that the Protocol has not yet been ratified by most countries. For our purposes, the latter is not a real problem, as the NEC decision (at least) confirmed the targets for the EU member states.

The costs of the initial NEC proposal (NEC+) represents an upper margin for the WTP, as NEC+ was not agreed upon by the Council. Therefore, the upper margin for the 'marginal' WTP can be set to approximately **350 €/ha**, taking into account all the hectares of ecosystems all over Europe.

IV.2.3.2 Shadow prices for impacts on ecosystems from emissions of SO_2, NO_X and NH_3

In the last step of the analysis, step 3, we estimate the shadow prices (damage or external costs) for acidification and eutrophication per ton of SO_2 and NO_X emitted.

We need to integrate our estimate on the marginal WTP, as calculated in the previous paragraph, in the impact pathway approach in order to calculate the shadow prices. Therefore, we first need to calculate the marginal impacts in physical terms, i.e. number of hectares of ecosystems for which the critical loads have been exceeded per additional ton of SO_2, NO_X and NH_3 emitted.

At this moment, steps have been undertaken to quantify the physical impacts of SO_2 and NO_X on ecosystems on EU level but it is too early to integrate them in the NewExt project.

Once the shadow prices are calculated, this data can be used to compare energy technologies and fuel cycles, used in the EU. The figures are additive to the ExternE figures, but are best separated, as they reflect another approach.

Although detailed results based on the most recent critical loads data are not yet available, first evidence[3] suggests that, on average for EU 15, these impacts are unlikely to make a major contribution to the total damage cost, but may be significant for emissions from countries or regions with low impacts on human health and relative high impacts on ecosystems.

[3] First estimates are based on critical load data from literature but that are outdated and do not match with the newer UN-ECE dataset used for the support of the Gothenburg Protocol and the NEC Directive (Hettelingh, private communication).

It has to be noted that the figures cannot be used in cost-benefit analysis or policy advice related to protection of ecosystems, as they are based on these policies.

IV.2.4) Validation and comparison of results with methods and results from literature

In order to evaluate the impacts of acidification and eutrophication we had to choose a physical indictor representing the impact on ecosystems and a valuation tool to quantify these impacts in monetary terms. The *critical loads* concept, expressed in terms of hectares of ecosystems for which the critical loads are exceeded, has been used as the physical indicator of the impacts and the *standard price approach* has been used to estimate the revealed preferences of policy makers, which is taken as a proxy for the preferences of society. In this paragraph, our methods and results are compared with those of studies with similar objectives.

We compare the approach in NewExt with the following approaches

1. Impact assessment: Use of critical loads or other physical indicators?
   - Dynamic modelling
   - Exceedance of critical loads expressed in terms of equivalents
   - Alternatives for impact assessment on ecosystems: Eco indicator 99.
2. The economic valuation of impacts based on ecosystems functions approach.
3. Abatement costs per ton of pollutant as shadow prices.
4. Expert weighting: Eco-indicator 99

IV.2.4.1 Impact assessment: use of critical loads or other physical indicators?

The critical loads concept has proven to be a helpful tool in the preparation of different policy targets on air pollution. The concept also has its limitations that have to be borne in mind when using them.

Based on a scoping study on the valuation of air pollution effects on ecosystems by MacMillan *et al.* (2001) one can list the following limitations of the critical loads concept:

1. The degree of damage is not considered by the critical loads concept as it is based on exceeding or not exceeding a threshold indicating environmental damage or not.
2. Critical loads provide a measure of ecosystem sensitivity but it is poorly linked to the observed patterns of the effects.
3. Secondary impacts, such as interactions with climate, pests, disease and multi-pollutant interactions with ozone are poorly represented in the critical loads concept.
4. The critical loads concept is mainly based on a steady-state, pollution-based criteria. Meeting the goal of not exceeding critical loads for a certain ecosystem does not imply the ‘recovery’ of this damaged ecosystem. So, there is no dynamic element within the critical loads approach, taking into account the past, actual and future status of acidification and eutrophication. Time delays and potential hysteresis during recovery phase are not assessed.

Dynamic modelling

In particular the latter limitation is important as ‘recovery’ of a damaged ecosystem is an important aspect of sustainable development of an ecosystem (Grennfelt P. *et al*., 2001; Posch M. and Hettelingh J.-P., 2001). Thus, where available, dynamic models should be used in further studies so that impacts on ecosystems can be evaluated in a long-term perspective.

Exceedance of critical loads expressed in terms of equivalents instead of number of hectares

Not only the use of the critical loads concept can be questioned. Critical loads can be expressed in different ways and each of these ways has its own characteristics. We used ‘the number of hectares of ecosystem for which the critical loads are exceeded’ to express the exceedance of critical loads in our determination of the WTP. We have chosen to do so because the goals of the policy targets (Gothenburg Protocol and NEC directive) were expressed as such and the marginal impacts of emissions on ecosystems are expressed in terms of hectares affected ecosystem per ton emitted pollution.
The exceedance of critical loads can also be expressed in terms of total accumulated exceedance (AE) of critical loads, expressed in equivalents deposition per year.
The AE of critical loads is calculated by multiplying the equivalents of deposition exceeding the critical load for acidification or eutrophication in a specific ecosystem (equivalents deposition/ha*year) with the number of hectares of that specific ecosystem affected (ha) and that summed over all the ecosystems to yield the accumulated exceedance for the whole of the grid cell.
In addition, by dividing the AE by the total ecosystem area of the grid cell, the average accumulated exceedance (AAE) (expressed in terms of equivalents deposition/ha*year) can be calculated.
The advantage of the AE and AAE is that it varies smoothly when deposition are varied and thus that it is not vulnerable for discontinuous distributions of the critical loads in a grid cell.
It would be interesting to use the accumulated exceedance of critical loads as the unit for the marginal impacts (equivalents deposition/ton pollutant emitted*year) and the WTP (€/equivalents deposition*year).

Alternative physical indicator for impact assessment on ecosystems: Eco indicator 99

An alternative physical indicator, based on the Potential Disappeared Fraction of Species (PDF) concept, has been used in the ‘Eco-indicator 99’ by PRé Consultants BV (Goedkoop M and Spriensma R., 1999). As this indicator is linked with a valuation step based on expert judgement, this is further discussed in paragraph IV.2.4.4.

IV.2.4.2 The ecosystems functions approach

From an economic point of view, the best approach is the valuation of the impact on the different functions of ecosystems. (regulation functions (e.g. CO_2 storage), recreation, information…). As it is strongly interlinked with the step of economic valuation, this is further discussed in the following paragraph.

Economic valuation of impacts on ecosystem functions

There are a lot of studies from North America and Scandinavia on the valuation of impacts of air pollution, and more in particular acidification. These studies are however not useful to validate our estimates. Comparison is hampered by the fact that we estimate figures which are different from most other studies:

1. We have calculated a WTP for the improvement of the health of the ecosystems, expressed per hectare of ecosystem that, thanks to an emission reduction program, does not longer exceed the critical loads for acidification and eutrophication. We do not make a distinction between effects on terrestrial ecosystems (forests, heath land, grassland) and aquatic ecosystems (freshwater).
2. Our scope is the determination of the shadow prices on national or EU15 level and not on local level. Figures determined for a particular region or country can not be extrapolated to other countries or scaled up for the whole of the EU15.

3. We did not focus on the values of the different functions of ecosystems: *users* values for production, information and regulation functions and *non-users* values as option, existence and bequest functions. The WTP we have calculated represents the total economic value.
4. The calculation of each of these values demand the application of the proper valuation tools (determination of stated, revealed or imputed preferences). We use the standard-price approach, which is not used for the determination of the values of the different ecosystem functions.

We have reviewed the most relevant and recent studies : Navrud S. (2002); Pearce D. and Howarth A. (2001); Ruijgrok E. *et al.* (2002), UN-ECE (2003); Wamelink *et al.* (2003); Witteveen+Bos (2001). In general, they show that :

- A wide range of tools is required for a full assessment of WTP for improvement of ecosystems using the *ecosystem functions* approach.
- The results available today are partial.
- A research strategy at European level is required to bridge all the gaps.
- Taking into account the differences between approaches, the data available cannot be used to validate our results.

IV.2.4.3 Abatement costs per ton of pollutant as shadow prices.

A number of studies use the abatement costs per ton of pollutant as directly as shadow prices and externalities for these pollutants. Examples include the study of Bernow *et al*. (1990) to assess external costs of energy, Greenstamp project, Pearce and Howarth (2001), Davidson D *et al.* (2002). These abatement costs can either be linked to a policy related target, e.g. Gothenburg protocol or NEC directive, or to scientific based targets (e.g. not exceeding critical loads).

At first sight, it looks attractive to use these shadow prices, e.g. for comparing energy fuels and technologies. They inform the decision maker that – if a certain technology with higher emissions is chosen – somewhere else additional expenses will be required in order to meet the policy or scientific emission reduction targets. The main advantage of this approach is that these single figures covers all impact categories, so that it is relatively simple to execute, compared to the damage function approach. The major assumption is that the targets include a careful balancing of costs and benefits. When the abatement costs reflect a policy target, this assumption may be justified by assuming that all pro's and cons have been taken into account in the decision making process. When it reflects a scientific target, this assumption is not valid.

From our point of view, the major drawback of these studies is that they may be a substitute for ExternE numbers, but cannot be directly combined with ExternE numbers for impacts on public health, agriculture and materials.

IV.2.4.4 Expert weighting: Eco-indicator 99

Eco-indicator 99 is an effect oriented method for Life Cycle Impact Assessment.[4] Next to the quantification of damage to human health and damage to mineral and fossil resources, the model also quantifies the damage to ecosystems. For acidification and eutrophication, the Potential Disappeared Fraction of Species (PDF) concept is used. As acidification and eutrophication do not always cause a decrease in species but rather a shift in species composition, only target species are included. These species are vascular plants, typical and representative for a certain ecosystem. The physical indicator used to quantify the damage to ecosystem quality is expressed as PDF * area * time [m^2*yr]. The effect of a change in SO_2, NO_X and NH_3 emissions on ecosystem quality is modelled by Natuurplanner, developed by RIVM and including a 250 x 250 m grid for the Netherlands. Natuurplanner is based on a soil model (SMART) and a vegetation response model (MOVE) and calculates the PDF*m*yr for over 40 types of ecosystems in the Netherlands taking into account the characteristics and background levels of each grid cell. (Goedkoop M. and Spriensma R., 1999)

This indicator can be further weighted. To this purpose, the Eco-Indicator has tried to estimate the total impacts of current pollution levels on ecosystems in one indicator, and has done the same for impacts on human health (DALYs), as well as all impacts related to resource depletion. Consequently, these three categories of impacts have been weighted using expert judgment. This system allows to compare impacts on ecosystems and human health for both SO_2 and NO_X emissions. As the impacts on human health can be valued, one can use this relationship also to value the impacts on ecosystems (De Nocker, 2001)

The Eco-Indicator is a useful tool for LCA studies. The embodied LCIA database allows to compare impacts from SO_2 and NO_X on ecosystems and on public health.

As indicated in paragraph IV.2.2, economic valuation should be based on the preferences of individuals, not experts. It has been argued that a big group of experts would reflect the different opinions and preferences in society, but there is no guarantee that this is really the case. In any case, the weighting of the Eco-Indicator group of experts is not confirmed by other studies (Cofala *et al.*, 2001). Basically, the outcome of the weighting process used in the Eco-Indicator is that impacts on human health and ecosystems got an equal weighting, and both are considered to be more (twice as) important as resource depletion. Our interpretation of these results is that experts say that they have no arguments to give a higher weighting to the total impacts on ecosystems, compared to the total impacts on public health. As a result, the whole weighting process is dominated by the extent to which impacts can be quantified.

IV.2.5) Priorities for further research

IV.2.5.1 Improvements of shadow prices based on implicit values of policy makers

The different steps of our approach can be improved, and will need to be updated as new information becomes available:

- There is not much information about the driving forces and mechanisms of the negotiations of the emission reduction programs. Therefore, important assumptions have

[4] More information about the use of eco-indicator 99 and the Natuurplanner model of the RIVM can respectively be found on the websites http://www.pre.nl/eco-indicator99/default.htm and http://arch.rivm.nl/milieu/natuurplanner/main.html (website in Dutch).

been made based on the analysis of official and legal texts of the policy targets and the associated documents on the targets settings and related costs and benefits. Further detailed information on the perception of key players involved in national and international environmental air pollution policy is crucial.

- Our approach refers to the discussion on the Gothenburg protocol and NEC directive, but it is to be expected that new policy targets for emission reductions will be set in the context of the CAFE program of the EC. This approach will include a wider multi-pollutant multi-effect approach, including impacts of secondary particles and PM in general on public health. In this context, our assumptions related to the use of data on public health damages will no longer be valid.
- Updated information on the marginal physical impacts of SO_2, NO_X and NH_3 emissions on ecosystems (in terms of change in exceedance of CL per ton emitted) should be plugged into the impact pathway in order to calculate the shadow price for impacts on ecosystems.
- The WTP has been determined in terms of € per *hectare* of ecosystem for which the CL has been exceeded. The 'implicit value of policy makers' approach demands the expression of CL in terms of hectares because the goals during the negotiations have been defined as such. Nevertheless, it would be useful to extract a WTP in terms of *accumulated exceedance of CL* from the one in terms of hectares, as it offers a more solid base in further valuation studies on acidification and eutrophication. Second, where available (e.g. CAFE, DG Environment), dynamic models should be used in further studies so that impacts on ecosystems can be evaluated in a long-term 'sustainable' perspective, including recovery aspects.

IV.2.5.2 Economic valuation of impacts on ecosystem functions

Our results - based on the *standard price* approach - can be used as complementary data to the ExternE numbers (based on the *damage cost* approach) to compare energy technologies and fuel cycles. However, research should continue to try to apply the damage function approach for impacts on ecosystems. To this purpose, the *ecosystem functions* approach needs to be further explored, building further on the results of the available studies (*cf.* report of the UN-ECE workshop (UN-ECE 2003)). This will require some more case studies to select priority impacts, and some applications of proven methodologies in order to get more data. These studies should be developed in a way that allows for benefit transfer in a second stage. The application of CVM questionnaires in a representative set of countries is an important step.
This approach is likely to result in a more complex set of data, with figures for valuation of ecosystem functions that may differ between ecosystems and countries. A major challenge will be to integrate the scientific information in this valuation framework. It is unlikely that the approach will be able to discriminate between emission reduction scenarios, as currently being discussed. Therefore, these studies should be complemented with development of application rules for policy oriented studies.

IV.3) Valuation of global warming impacts based on implicit values of policy makers

IV.3.1) Context and objectives

Externalities estimated based on the impact pathway approach in ExternE resulted in a best estimate ranging from 0.1 - 16 €/ton of CO_2-eq. (Tol and Downing, 2000) A review of recent literature (2003) confirms this general picture and data, although the range of data in literature is wider, with some studies offering much higher estimates. (Defra, 2003) However, it remains unclear to which extent these data give a complete picture of the total impact, as a wide number of impacts are not included and for those that are included, uncertainties are large, both for quantification of effects as for the valuation.

Given the uncertainties and incompleteness inherent to these estimates, one can argue that the balancing of costs and benefits in negotiations over targets and/or policy measures may offer a complementary view on how society values the benefits of the first steps in CO_2 control. Therefore, in NewExt two approaches based on revealed preferences have been explored. The first is to estimate a revealed preferences based on policy targets. A second approach is based on public preferences as revealed in referenda related to energy questions in Switzerland. The latter is discussed in paragraph IV.4.

IV.3.2) Selection of policy targets and their interpretation

To estimate the revealed preferences, similar information and data are required as for revealed WTP to limit impacts from acidification on ecosystems, and similar steps in the analysis are required. A first issue is to selects the most relevant policy targets, and to interpret the arguments used in the negotiations leading up to these decisions.

The main target at EU level is the Kyoto protocol of 1997, which has been ratified by the EU and its member states in 2002. The European Climate Change Program of 2000 elaborates a roadmap to translate this target into proposals. [5]

The Kyoto protocol defines the target for the EU to reduce greenhouse gas emissions by 8 % by 2008-2012 compared to 1990 emissions, for the EU 15 as a whole. The protocol itself however does not indicate how the target should be achieved. This is an important question because the costs of meeting Kyoto will depend on the policy mechanism chosen.

The policies in Europe related to climate change show a tradition of looking for a balance between

- dividing the target between member states and sectors, leaving it open to member states and/or sectors to look for measures how to achieve these targets,
- deciding at EU level on concrete policy measures, sector specific or cross sector. (e.g. a CO2 tax or EU wide emission trading system)

The final decisions still show a mixture of these approaches:

- First, the EU has developed differentiated targets for each member country in order to share equitably the economic burden of climate protection. This so called "burden-

[5] Although the protocol has not yet entered into force (at the date of the latest update of this report in August 2004) this does not affect our interpretation of the commitment of the EU to control the emissions of greenhouse gasses.

sharing" agreement between EU governments lays down differentiated emissions limits for each Member State with the aim of ensuring that the EU meets its overall 8% reduction commitment under the Protocol. The limits are expressed in terms of percentages by which Member States must reduce, or in some cases may hold or increase, their emissions compared with the base year level (1990). The national commitments are shown in Table .These differentiated targets for countries reflects that the costs and the capacities to carry these costs may differ, as well as society's willingness to take early action. The EU member states have to develop National Allocation Plans (NAPs) to indicate how they will achieve these emissions reductions.

- Second, a combination of measures at European and national level is required, including flexible mechanisms like the EU Emissions Trading Scheme (EU ETS) due to start on the first of January 2005. The objective of the latter is to allow for a cost efficient reduction of CO_2-emissions for big industrial energy users. In addition, additional measures and targets will be required, e.g. for transportation and household sectors, both at European and national level.
- Third, countries have the possibilities to meet their emission reduction targets by using the so-called flexible mechanisms or Kyoto Mechanisms, like Joint implementation, International Emissions Trading, etc.)

IV.3.3) A shadow price for CO_2 emissions in Europe

Towards the preparation of the Kyoto protocol, the potential for CO_2 emission reduction in the EU and their costs were well documented. Therefore, it is fair to say that in preparing and implementing Kyoto agreement, these costs were balanced against the benefits. There are several limitations for the use of this information as a revealed preference from policy decisions.

- First, the real preferences will be revealed in the policies implemented, rather then in the phase of setting targets. This would however require a careful assessment of all national plans to see which policy measures will be implemented, to see the real 'willingness to pay to combat global warming' from policy and decision makers. This work can only be done when the final plans are available and accepted by the EC. Therefore, this analysis is based on more generic information on reduction costs per ton of CO_2. For the interpretation of the data, we will use some additional information on policy plans etc.
- Second, the main benefit of the first steps towards CO_2 control is not only a reduction in damages from global warming, but they also contribute to build a world-wide strategy to combat global warming. In this context, the benefits of meeting the Kyoto target (expressed per ton of CO_2) may have a multiplicator effect, which is not reflected in the figures used for the decision making/
- Third, controlling CO_2 emissions will result in benefits in other areas including air quality, energy security, etc. These so called no-regret benefits have not been documented in detail and are not accounted for.
- The discussion and data mainly focused on one GHG, i.e. CO_2, whereas the protocol covers all greenhouse gases.

IV.3.4) A shadow price for CO_2 emissions in Europe

In the policy process leading to the adaptation of the European Climate Change Program and the proposal for a directive on CO_2 trading mechanism, several studies on the costs of meeting these targets were executed, mostly using energy-economic models. The latest studies for the EU suggest that under a full flexibility EU-wide allocation of least cost sectoral objectives, the marginal abatement cost will be 20 euro per ton. These estimates are based both on top-down and bottom up approaches. A recent review showed that this estimate is in the middle of the wider range of estimates, both from studies and from starting or experimental CO_2-trading schemes. (Downing and Watkiss, 2003). When however each member state will try to fulfill their objectives on their own, the marginal cost for Belgium will increase up to 90 € per ton CO_2 (Blok, 2001). On the other hand, allowing some kind of trading outside the EU may lower the compliance costs to perhaps 5 € per ton. Consequently, most studies take a figure close to this 20 € per ton of CO_2 as the marginal abatement costs, and a proxy for the society's willingness to pay, for Europe. This number is also well below the penalty set in the emission trading scheme (40 € per ton of CO_2 for the first 3 years),and which can be seen as an upper limit for this shadow price.

From a theoretical point of view, there are reasons to argue for higher or lower numbers, but our analysis shows that they are no better estimates then the range of 5-20 €/ton CO_2.

- As a number of countries accepted stricter emission reduction targets and took earlier unilateral actions to limit CO_2 emissions, and as studies indicated that they would also require the more costly emission reductions, one can argue that the WTP in some countries may be higher. Given the differences in emission reduction targets, and the costs to meet these targets, there are good reasons to argue for country specific shadow prices for CO_2. Consequently, some propose a national shadow price for CO_2. As an example, from analysis of policy targets for the Netherlands and national costs estimates, a shadow price of 50 € per ton of CO_2 equivalent is proposed. (Davidson et al, 2002)
- Although the marginal abatement costs for reaching the objectives are available per country, these cannot be taken as a proxy for society's WTP per country, unless more evidence to support such values is available. One may also argue that for many member states their recent record in emission trends does not support the idea of a high WTP, as most member states lag behind a theoretical linear Kyoto target path. (EEA, 2004) Second, a recent overview of draft national plans illustrated that a number of countries will need the cheaper Kyoto flexible mechanisms to reach Kyoto target. (Ecofys, 2004). The costs of using flexible mechanisms will be lower, but it is still unclear to which extent these mechanisms will be used and what the marginal prices are likely to be.
- One can argue that the market prices for CO_2 emission allowances under the EU ETS inform us about the real 'shadow price' for CO_2 and the real WTP from policy makers.
- It is hard to estimate to which extent a shadow price will be reflected in real life decision making in the sectors because it is very unclear to all potential actors in the market how this market will develop.
- Indeed, as the industries subject to the EU ETS will receive emission allowances (grandfathering) based on the national allocation plans, national governments will make some cost-benefit considerations in controlling CO_2 emissions in sectors subject to the ETS or in other sectors. It is unlikely that a future market price for CO_2 emission allowances also has been taken into consideration, because this development of this market and future prices are very unclear. In the long run however, if the EU

ETS scheme develops into a real market, this could be a better indicator then current data from techno-economic studies.

- It may be argued that the real WTP will be lower then the range suggested above, because policy makers are aware of benefits in other areas like energy saving or air pollution. Although the argument is true, there are no data to correct for this potential effect. This remark is in support of choosing a best estimate in the lower side of the range.

IV.3.5) Application of shadow prices for CO_2 and greenhouse gases

An assessment of the costs for achieving Kyoto targets can be interpreted as a proxy for society's willingness-to-pay for early action against global warming. For assessing technologies and fuel cycles in the mid-long term, the best estimate is between 5-20 €/ton of CO_2, with the higher range reflecting the costs if emissions are controlled within Europe. By extension, it can be applied to all greenhouse gases. For application in New-Ext case studies, a value of €19 / ton CO_{2equi}. has been selected.

This shadow price for CO_2, based on the marginal abatement costs to meet the Kyoto target, reflects the CO_2 efficiency of energy technologies or fuel cycles. Those that are more efficient will be given credit for this benefit, which allow European society and economies to save costs for meeting the Kyoto target.

When applying this range, some remarks have to be considered. First, it needs to be evaluated on a case by case base whether this figure is applicable and whether some kind of CO_2-externality has already been internalized. Within the sectors subject to the emission trading regime (e.g. electricity generation), a price incentive that reflects CO_2-efficiency will be installed from 2005 onwards. A-priory, one cannot decide however to which extent the EU ETS scheme will develop into an active market.

The average electricity price for consumers, however, will not contain a price signal that reflects overall CO_2 efficiency. When comparing technologies on a full fuel cycle base, emissions outside the EU are unlikely subject to price incentives that reflect CO_2-efficiency.

Second, depending on the context, a sector or country specific marginal abatement costs may be better than the European marginal abatement cost. This is the case if the shadow price needs to reflect the contribution of that technology or fuel cycle to a specific target at national or sectoral level. This will be especially the case for decisions with a short time impact, and limited to a specific sector or country. The same reasoning goes for shadow prices for other greenhouse gases. On the other hand, if the objective is to reflect some overall shadow price for making (small) progress towards controlling greenhouse gases, the overall marginal European marginal abatement cost for CO_2 is a better proxy, and can be applied to all greenhouse gases. This will especially be the case for decisions with a longer time horizon, and a cross-sector or cross-border impact.

IV.4) Public preferences for CO_2 control revealed in referenda in Switzerland [6]

An innovative approach was developed by deriving an implicit Willingness-To-Pay (WTP) for controlling CO_2 emissions from people's voting behaviour in referenda related to energy questions in Switzerland. Decision making in Switzerland differs essentially from decision making in other countries due to strong components of "direct democracy". In many cases, key Swiss policy issues are decided by a national referendum. There have been a number of Swiss national referenda related to the subjects "energy" and "environment". Some included decisions about prices/taxes. Referenda can be viewed as large surveys, which at the same time constitute political decisions.

The referendum method promises some advantages compared to the usual survey methods for a couple of reasons.
- The decision is not fictitious but "serious", i.e. if the population would decide to introduce a certain tax it would have to be paid really.
- It is clear to the individual that he or she is not the only one who has to pay. Instead, in case the decision would be accepted, the whole population would be involved.
- Because of the large number of people involved, a referendum is much more representative than a usual survey. About two million voters participated in each of the referenda considered here.

On the other hand, the referendum method has also disadvantages and limitations:
- Surveys are more flexible and can be more detailed.
- It is practically impossible for researchers to initiate a referendum and to determine the "questions"; the existing formulation of the referendum text has to be taken as it is.

The referenda provide the unique chance to study the revealed opinion of a huge number of persons. There have been a couple of Swiss national referenda related to the subjects "energy" and "environment". Some included decisions about prices/taxes. The idea of the proposed method is to use results from referenda related to environmental issues to estimate preferences of the population.

IV.4.1) Methodology

The general problem is that a referendum asks only for a yes-no decision. So one has only the number of yes-no answers for the whole country or for some subgroups (e.g. cantons in Switzerland). From these yes-no results, conclusions have to be drawn somehow about the underlying preferences.

The basis for the extraction of the underlying preferences is the following assumption:
"Asked about the payment of a certain price p, people who have a Willingness-To-Pay (WTP) equal to p or higher than p vote yes and people with WTP lower than p vote no. Then the share Y of yes-votes in percent gives the (100-Y)-percentile for the price p for the underlying statistical distribution of the WTPs of the individuals. "

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6 This part is based on: Heck Thomas, Referendums in Switzerland - revealed preferences related to energy and environment, Paul Scherrer Institut, 2003

The remaining issues are:

- This applies in a strict sense only to the conditions of the full referendum text i.e. including decision details about intended purposes etc. Further assumptions about the influence of the formulation on the voters' decisions and about the transferability of results are necessary (e.g. question about price per kWh, transfer to price per ton CO_2). Fortunately, for the application to CO_2 in NewExt, some reasonable assumptions can be made on interpretation of the results.
- The total outcome of a single referendum does not fix the details of the statistical distribution. Fortunately, for the application to CO_2 in NewExt, the results of several referenda can be used.

The mathematical description to derive a WTP of individuals has been developed and is elaborated in more detail in Heck (2003).

It has to be noted that the results refer to the preferences of Swiss people, and, as for all results of valuation studies, cannot be simply transferred to other countries. However, because of the large amount of data and results per canton, some elements important for benefit transfer can be further studied. First, issues like impact on unemployment rate, urban/rural structure, industrial structure or other economic conditions can have influence on the WTP for environmental protection. Cultural differences are a general problem for the transfer of results on preferences/WTP to other countries. To a certain extent the influence of cultural differences to decisions about referenda can be estimated. Differences can be observed between parts of Switzerland which are closely related to their near neighbouring countries (France, Italy, Germany, Austria). By investigating these differences it may be possible to extrapolate the results approximately to other central European countries.

IV.4.2) Results

IV.4.2.1 Selection of relevant referenda

A list of national referenda related to energy or environment is discussed in Heck (2003). The most important recent referenda, which can be used to deduce concrete numbers on revealed WTP in Switzerland, are the following (Confoederatio Helvetica, 2002):

1.) September 24, 2000: "Solar-Rappen/Solar-Initiative".
2.) September 24, 2000: "Förderabgabe für erneuerbare Energien" = Gegenentwurf der Bundesversammlung zur Solar-Initiative.
3.) September 24, 2000: "Energielenkungsabgabe", Gegenentwurf der Bundesversammlung zur zurückgezogenen Energie-Umwelt-Initiative:

The referenda above all have been rejected by the majority of the voters. The results show clear differences in the different cantons of Switzerland. Some energy-related initiatives, although rejected in the whole country, reached a majority in some cantons. Some initiatives came close to the majority level. So the proposed numbers are probably not so far away from the average WTP threshold of the population.

Two of the referenda suggested a single concrete price and mentioned comparable intended purposes. The results and the queried prices per kWh are shown in Table IV-4.

Table IV-4 : Referendum results and suggested price per kWh.

	Price [CHF/kWh]	Intended purpose	Remark	% Yes
Solar-Initiative	0.005	A	Final price	32%
Förderabgabe (Gegenentwurf zu Solar-Initiative)	0.003	A’	Fixed price	47%
Energielenkungsabgabe	0.02	B	Maximum price	45%

Intended purposes: A and A’ to renewable energy only, B for reduction of social costs also.

The results for A and A’ are consistent with the assumption that for similar intended purposes of the proposed tax mainly the price is relevant for the decision, i.e. the lower the price the higher the acceptance. This was also true for all results in the single cantons without any exception. It was also true for the relation to political parties according to surveys (Ballmer-Cao et al. 2003).

IV.4.2.2 Analysis of results of a single referendum

Figure IV-5 shows the statistical distribution of the votes over the 27 canton results. Altogether 44% of the holders of the voting rights (2'090'548 of 4'676'509) participated in the referendum. Similar analysis has been performed for all selected referenda.

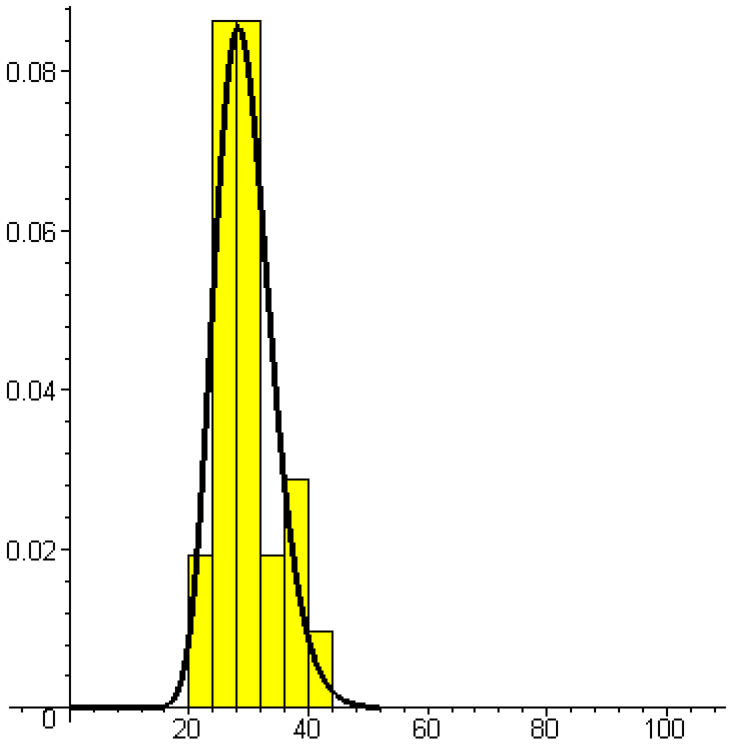

Figure IV-5 : Distribution of yes-votes [in %] for Solar-Initiative in the different cantons (y-axis normalized to 1)

IV.4.2.3 Derivation of WTP results

The analysis of the statistics of the referendum results (based on the two referenda with comparable intended purposes, i.e. "Solar-Initiative" and "Gegenentwurf zu Solar-Initiative") yields for a suspected underlying WTP (see section "Mathematical description" in Heck 2003):

Table IV-5 : Result for estimated Willingness-To-Pay in Switzerland based on referenda

	Willingness-To-Pay
Geometric mean/Median:	0.00272 CHF/kWh
Arithmetic mean	0.0064 CHF/kWh
Geometric standard deviation:	3.7

IV.4.2.4 Conversion from results per kWh to results per ton CO_2:

The referendum text refers to final energy (i.e. sold to customer) of non-renewables (Solar-Initiative) or energy content of non-renewables (Gegenentwurf).

For simplicity, the calculated WTP is accounted only to CO_2 emissions. Arguments for accounting to CO_2 only are:

- The change from fossil fuels to renewable energy affects mainly direct CO_2 emissions but not necessarily other pollutant emissions (e.g. NO_x or PM_{10} emission factors for biomass are comparable to those for fossil fuels).
- Other emissions than CO_2 are relatively low in Switzerland.

An average emission factor for non-renewable energy in Switzerland of about 230 g/kWh has been estimated (Heck, 2003). Thus the constructed WTP result above corresponds to:

Result for estimated Willingness-To-Pay (CO_2) in Switzerland based on referenda:

Geometric mean/Median:	about 12 CHF/ton CO_2	(about 8 Euro/ton CO_2)
Arithmetic mean:	about 28 CHF/ton CO_2	(about 19 Euro/ton CO_2)

It was assumed that the WTP per kWh is fully accounted to CO_2 and the differences in the proposed intended purposes of the tax are not extremely relevant. Prices refer to year 2000.

IV.4.3) Conclusions on valuation based on pubic referenda with application to CO_2.

Within NewExt, results of referenda on energy taxes held in year 2000 have been analyzed. Under plausible assumptions about the underlying WTP distribution, the average willingness of the Swiss population to pay energy taxes per kWh can be estimated. The referenda originally refer to taxes on non-renewable energy consumption in order to favour renewable energy. The change from fossil fuels to renewable energy affects mainly direct CO_2 emissions but not necessarily other pollutant emissions (e.g. NO_x or PM_{10} emission factors for biomass are comparable to those for fossil fuels). Therefore it is plausible to account the WTP per kWh fully to CO_2 as far as emissions are concerned.

The resulting estimates are about 6 to 9 €/ton CO_2 for the geometric mean and about 14 to 22 €/ton CO_2 for the arithmetic mean.

This estimate is of the same order of magnitude as the one derived on literature on cost-efficient implementation strategies to meet the Kyoto protocol. The estimated WTP is however significantly lower than the abatement costs in Switzerland (starting at about 100 CHF/ton CO_2 i.e. about 70 Euro/ton CO_2).

IV.5) Overall conclusions

IV.5.1) Interpretation of policy decisions and referenda to derive a willingness to pay

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

The analysis shows that a simple analysis may not be correct, i.e. abatement costs for SO_2 and NO_X need to be corrected for other impacts and incentives to internalise CO_2 shadow prices need to be checked carefully.

By analyzing in detail the decisions of policy makers and in addition public referenda, shadow prices for global warming (ca. 5 to 22 € per ton of CO_2) and exceedance of critical loads for eutrophication and acidification (ca. 100 € per hectare of exceeded area and year with a range of 60 – 350 €/ha year) have been developed.

IV.5.2) On the use of the numbers derived in this chapter

The proposed shadow prices for impacts on ecosystems are additive to other ExternE impact figures. The shadow prices for CO_2 are not additive, but rather offer a complementary perspective. These figures are best separated from estimates of damages.

These figures can be used for comparison of technologies and fuel cycles, similar to e.g. life cycle impact assessment tools, like Eco-Indicator. They cannot be used to evaluate environmental policy objectives related to these pollutants or objectives.

The evaluation has also shown that under certain assumptions, results of referenda related to energy questions can be interpreted as revealed preferences to tackle environmental problems. A first exercise for Switzerland shows that individual preferences to control CO_2 emissions may be of the same order of magnitude as marginal abatement costs for the EU.

IV.5.3) Recommendations for further research

It is recommended that further studies should focus on up to date data for marginal impacts of emissions on critical loads, in order to have up to date data.

It is recommended that the current data and interpretation schemes are kept up to date with the policy developments related to long range transboundary pollution and especially climate change.

It is recommended that the approach to extract revealed preferences from referenda is further explored, so that the results of such referenda can be used in a broader policy context.

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V ASSESSMENT OF ENVIRONMENTAL IMPACTS AND RESULTING EXTERNALITIES FROM MULTI-MEDIA (AIR/WATER/SOIL) IMPACT PATHWAYS

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Summary

The goal of this work package is to develop a site dependent model for the assessment of external costs from priority impact pathways via soil and water, and to apply it to the emission of toxic substances by power plants. Of particular concern are the toxic metals As, Cd, Cr, Hg, Ni and Pb; for these we have calculated the collective dose. At the start of this work the available models were reviewed, with the conclusion that none of them can be used directly for this purpose. However, by suitable modifications and adaptations we have been able to develop two independent models, and we have calculated results for the collective dose per kg of emitted pollutant. One of the models (the "Uniform World Model”) is based on transfer factors and other parameters of EPA, the other (“WATSON”) is a multi-zonal model that links the regional air quality model of EcoSense to a soil and water multi-media model of the Mackay level III/IV type. Dose-response functions have been reviewed. Impacts and damage costs have been quantified for As, Cd, Cr, Ni and Pb.

The output of these models is the damage per kg of pollutant, as a function of the site and conditions (for emissions to air: stack height, exhaust temperature and velocity) of the source. In the present report only emissions to air are considered; direct emissions to soil or water could readily be evaluated when data for such emissions become available. The emissions, per kWh, of toxic metals by coal and oil fired power plants are estimated, to obtain the resulting contributions to the cost per kWh; they turn out to be very small.

V.1. Introduction

The goal of this work package is to develop a site dependent model for the assessment of external costs from priority impact pathways via soil and water, and to apply it to the emission of toxic substances by power plants. Of particular concern are the toxic metals As, Cd, Cr, Hg, Ni and Pb, as well as certain organic pollutants, in particular dioxins. The output of this model is the damage per kg of pollutant, as a function of the site and conditions (for emissions to air: stack height, exhaust temperature and velocity) of the source. In the present report only emissions to air are considered; direct emissions to soil or water could readily be evaluated when data for such emissions become available. The emissions, per kWh, of toxic metals by coal and oil fired power plants are estimated, to obtain the resulting contributions to the cost per kWh.

At the start of this work several existing models for the calculation of doses have been considered in detail, in particular EUSES [1997], CalTOX [McKone & Enoch 2002], the model of EPA [1998a] for waste incineration, the model of IAEA [2001] for radionuclides, and the Vlier-humaan (VH) multi-media exposure model of VITO. None of these models can be used directly for the calculation of external costs because they do not quantify the total impact of an emitted pollutant but only the impact in a limited region, over a limited time horizon or on a limited population (the most exposed subgroup). Since the external cost should take into account the total impact (expectation value rather than worst case estimate), over all time, all space and the entire population, these models have to be adapted.

We have, therefore, decided to develop two new models, based on elements of the above models. The first model, called “uniform world model” (UWM) [Spadaro & Rabl 2003] is based mostly on EPA [1998a], with some supplemental data of IAEA [2001]; in its present version it focuses on toxic metals (As, Cd, Cr, Hg, Ni, and Pb) because these are the most troubling emissions of the energy sector. The second model, called WATSON, is an extension of the existing EcoSense model [European Commission, 1999] by the integrated WATer and SOil environmental fate, exposure and impact assessment model of Noxious substances for Europe [Bachmann, 2003]. It is a multi-zonal model that links the regional air quality model of EcoSense to a soil and water multi-media model of the Mackay level III/IV type.

In addition, the VH model of VITO has allowed us to carry out certain sensitivity studies to get a sense of the reliability of the results of UWM and WATSON. A model like VH, aimed at deriving soil standards, and based on specific land use types, is by concept totally different from UWM and EcoSense/WATSON. There should, however, be consistency when looking at ratios of ingestion to inhalation. These tests with VH are reported in Section V.4.

Finally, to obtain damage cost estimates, one also needs the dose-response functions (DRF) or concentration-response functions (CRF), as well as unit costs of the corresponding end points for the monetary valuation. Here a crucial limitation lies in the paucity of available information. For most substances and non-cancer impacts the only available information covers thresholds, typically the NOAEL (no observed adverse effect level) or LOAEL (lowest observed adverse effect level). Knowing thresholds is not sufficient for quantifying impacts; it only provides an answer to the question whether or not there is a risk. The principal exceptions are carcinogens and the classical air pollutants, for which explicit dose-response functions are known (often on the assumption of linearity). For other substances and end points one could apply recent work on estimating DRFs [Pennington et al 2002]. We have found suitable DRFs for cancers due to As, Cd, Cr, and Ni, as well as for IQ decrement due to Pb, but so far we have not been able to quantify the damage cost due to Hg.

V.2. The "uniform world model" (UWM)

V.2.1. General Considerations

The starting point is the observation that for incremental impacts due to small (compared to background levels) changes in emissions the dose-response function (DRF) can be linearized and the corresponding total damage can be calculated with equilibrium models (steady state) even though the environment is never in equilibrium[1]. The necessary equations and parameters for the assessment of As, Cd, Cr, Hg, Ni and Pb are obtained from EPA [1998a]. The model is a generalization to multimedia of the "uniform world model" for air pollution of Curtiss & Rabl [1996] and Spadaro [1999]; it provides typical results for a region rather than for a specific site. Nonetheless it can distinguish, by means of simple correction factors, different kinds of sources such as power plants, industrial boilers and cars.

We account for the pathways in Fig.V.2.1. We do not consider dermal contact because that pathway has been found to be entirely negligible for these metals [e.g. EPA 1998a, McKone & Enoch 2002]. Like the underlying model of EPA [1998a] we do not consider ground water, assuming that on average inflow and outflow of the pollutant to this compartment are equal.

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

In the same spirit we assume that all drinking water is taken from surface water rather than groundwater. The resulting drinking water dose is an upper bound because it does not account for removal processes during the passage to and from groundwater.

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

The model is fully documented in Spadaro & Rabl [2003].

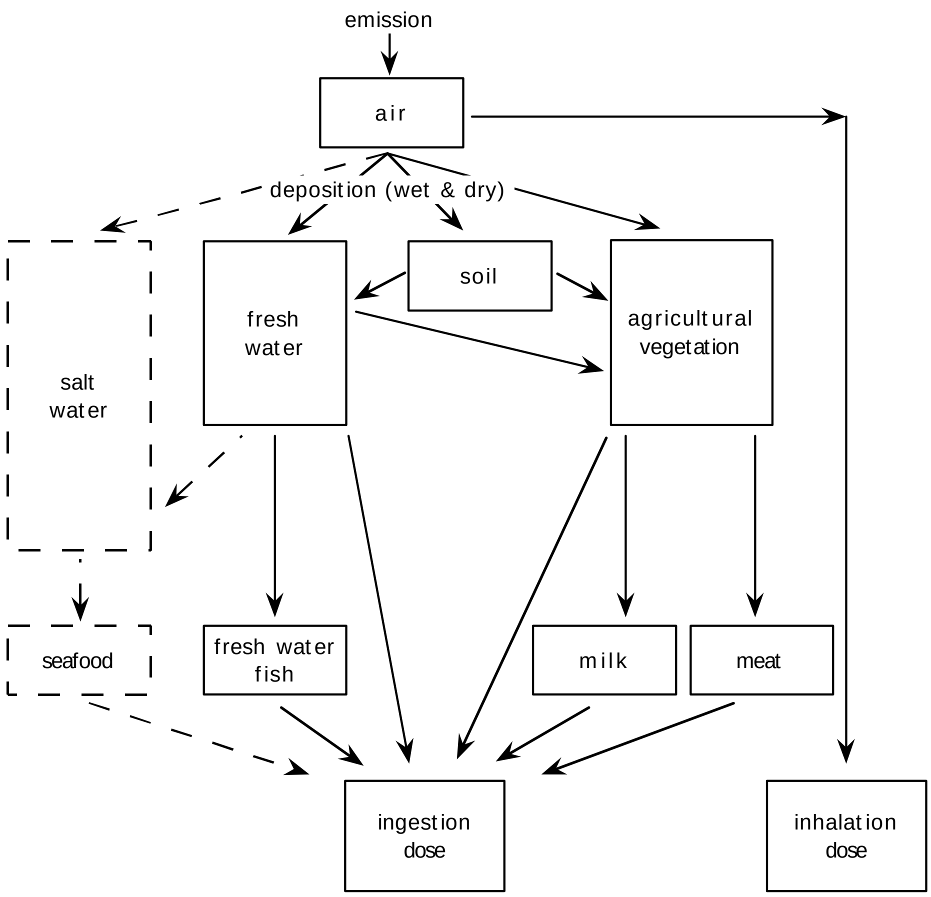


Fig.V.2.1. Pathways taken into account for health impacts of air pollutants. Direct emissions to soil or water are a special case where the analysis begins at the respective “soil” and “water” boxes. In the present version seafood is not yet included.

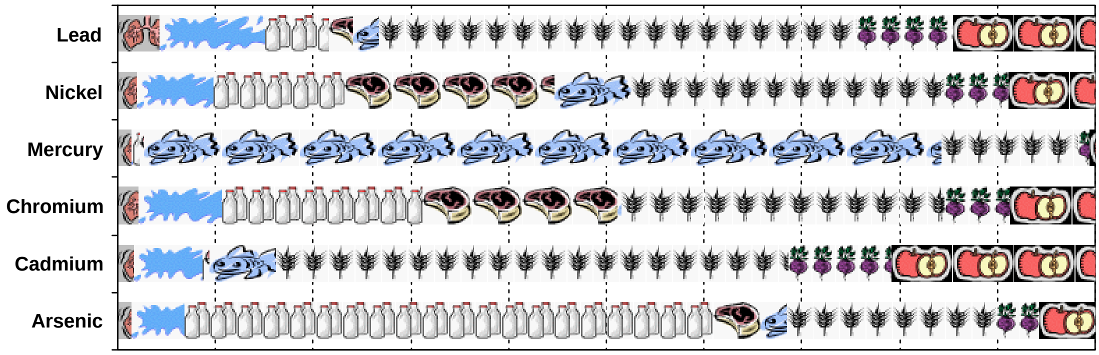
V.2.2. Results of UWM for Doses

Fig.V.2.2 shows the collective dose in mg due to the atmospheric emission of 1 kg of the respective metals under typical central European conditions. The average population density is 80 pers/km^2 (land and water). Taken as dimensionless ratios, the numbers in the table under Fig.V.2.2, multiplied by 10^{-6}, are the fraction of the emitted pollutant that passes through human bodies; this is sometimes called intake fraction.

The doses shown are the total ingested or inhaled quantities, without regard to the fraction that is actually absorbed. If the absorption rates are less than 100%, they have to be included before applying DRFs that are based on absorbed dose. Inhalation and ingestion can be associated with very different DRFs. For example, As is more carcinogenic, per mass, if inhaled than if ingested. Ingestion of methyl mercury is much more harmful than inhalation of Hg vapor, which in turn is more harmful than ingestion of elemental Hg. The ingestion doses for Hg in Fig.V.2.2 should be reduced by the fraction of Hg that is actually transformed into methyl mercury.

The total dose can be much larger than the inhalation dose, by about two orders of magnitude. A simple back-of-the-envelope calculation may be instructive to explain why ingestion can be so much more important than inhalation. Consider an average person exposed to air with concentration c_{air}. The annual inhalation dose is $D_{inhal} = c_{air}\ V_{inhal}$ with an annual inhalation volume $V_{inhal} = 7520$ m^3/(pers·yr) (we take a period of a year but that choice has no effect on the result since the argument involves time-averaged values). If aboveground food crops are exposed to the same concentration, the ingestion dose due to direct deposition on the plants is $D_{ing} = c_{air}\ v_{dep}\ A_{crop} \times 1$ yr where A_{crop} is the horizontal area of the crops intercepting the deposition flux. The plant yield of 2.24 kg$_{dW}$/m^2 (Table A3 of Appendix of Spadaro & Rabl [2003]) together with the consumption rate of 127 kg$_{DW}$/(pers·yr) for aboveground crops implies an area of 127/2.24 = 56.7 m^2 per person; however, this number has to be reduced by the fraction of the year the plants are grown (say 2 months/yr) and by the ratio intercepting area/ground area (say 0.2). Thus we take A_{crop} = 56.7 m^2 × (2/12) × 0.2 = 1.9 m^2. The resulting ratio of ingestion and inhalation doses is $D_{ing}/D_{inhal} = v_{dep}\ A_{crop} \times 1\ \text{yr}/V_{inhal} = 40$, for a typical v_{dep} of 0.005 m/s = 1.58E5 m/yr. The ratio is reduced to the extent that the pollutant is not absorbed by the edible portions of the plant, but it is increased by the contributions of belowground crops, milk and meat. In any case, this explains why for pollutants that are absorbed by plants, ingestion can indeed be a much more important than inhalation.

We have performed a sensitivity analysis to evaluate how much the intake fractions vary with changes in the input parameters; the results are shown in Table A3 of Appendix of Spadaro & Rabl [2003]. The most critical parameters, except for Hg, are the yield per planted area, and in some cases the soil-plant bioconcentration factors and the biotransfer factors for meat and milk. For Hg the most critical factor is the bioconcentration factor for fish. The choice of the deposition velocities is not very critical because of the relative smallness of inhalation. The last lines of Table A5 show how the doses change if the cutoff time t_{cut} in Eq.6 is changed to 30 yr and to infinity: extending the time horizon has an appreciable effect for Hg and Pb because of their long soil loss time constants.



	Arsenic	Cadmium	Chromium	Mercury	Nickel	Lead
TOTAL (Base case)	283.5	228.0	181.6	1546.5	200.2	164.1
Green vegetables	16.2	47.5	15.9	9.2	17.6	23.9
Root vegetables	12.4	24.1	12.0	16.3	13.1	15.9
Grains	60.6	119.3	60.2	217.1	64.2	80.2
Freshwater fish	7.8	15.8	0.8	1261.3	15.8	4.4
Cattle meat	13.6	1.3	36.1	7.4	43.1	4.1
Cattle milk	153.5	0.3	37.3	10.6	26.7	10.7
Water	15.6	15.8	15.5	2.5	15.8	17.8
Inhalation	3.9	3.9	3.9	22.0	3.9	7.1

Fig.V.2.2. Collective doses for central European conditions, by exposure pathway as a percentage of the total (figure) and in mg per emitted kg (table) for base case (t_{cut} = 100 years). Doses from seafood are not included. As, Cd, Cr and Ni are modeled as PM_{10} and Pb as $PM_{2.5}$; Hg is modeled as metallic Hg for inhalation, methyl Hg for ingestion. The inhalation doses are for typical power plant emissions (stack height around 100 m); they should be multiplied by about 3 for typical industrial emissions (near cities, stack height 0 to 40 m), and by about 20 for typical automotive emissions in cities. Doses from drinking water are lower if the water utilities remove toxic metals.

The results for the ratio ingestion/inhalation are consistent with data reported by WHO [1988 – 2001]. Among models that should give comparable results we have found the CalTOX model of McKone & Enoch [2002], a level IV model in the terminology of MacKay [2002]. It analyzes essentially the same pathways as UWM; in particular ingestion of seafood and exchanges with ground water are not considered. We have run CalTOX for the most comparable scenario, i.e. in the continuous emission mode with the settings landscape = US, start of exposure = 30 yr, exposure duration = 70 yr, and exposure factors = LCIA, to calculate the intake fractions for inhalation and for ingestion. Our inhalation doses tend to be higher than those of CalTOX, our ingestion doses lower. Table V.2.1 shows the ratios of the total doses calculated by UWM and by CalTOX, after multiplying the CalTOX results by the ratio 80/29 of population densities in the EU and the USA. For As, Cr and Ni the results are fairly close, considering the uncertainties; for Cd and Pb the UWM numbers are 14 to 20 times lower. We do not show a comparison for Hg because the differences in modeling are too large: UWM calculates the total dose in the entire hemisphere (inhalation in metallic form, ingestion as methyl Hg), whereas CalTOX considers only the dose in the USA (for Hg or for methyl Hg, without transformation from Hg to methyl Hg).

Table V.2.1. Ratio of total doses calculated by UWM and by CalTOX, after multiplying the CalTOX results by the ratio 80/29 of population densities in central Europe and the USA.

Ratio of doses	**As**	**Cd**	**Cr**	**Ni**	**Pb**
UWM/CalTOX	0.61	0.07	0.39	0.60	0.05

There are two principal differences between CalTOX and UWM: one lies in the modeling of atmospheric dispersion, the other in the transfer between compartments. Whereas UWM uses empirically determined transfer factors, mostly of EPA, CalTOX calculates the transfer by means of fugacity data. For the dispersion in the atmosphere we assume that As, Cd, Cr and Ni are emitted as part of PM_{10} (for industrial and power plant emissions) and Pb as part of $PM_{2.5}$ (for automotive emissions), being dispersed and deposited on the ground like other particulate matter of its size. CalTOX, by contrast, assumes emission in pure metallic form, the metals then attaching themselves to other particulate matter in the atmosphere according to the fugacity between the metal under consideration and the particles that are already in the atmosphere (their concentration is one of the input parameters). Since the transfer from metallic phase to particles occurs at different rates for different metals, the atmospheric residence time and hence the inhalation dose are different for different metals. The atmosphere of CalTOX is modeled as a homogeneous perfectly mixed compartment with volume equal to the height of the atmosphere times the impact area under consideration (land area of the USA for the setting landscape = US). We believe that the treatment of the atmosphere in UWM is more realistic because it has been explicitly validated by numerous comparisons with detailed atmospheric models. For the transfers between the other compartments we do not know which approach is more reliable.

V.2.3. Results for Impacts and Social Costs

Obviously impacts can be quantified only to the extent that the slopes of the CRFs or DRFs in the relevant dose range are known. Unfortunately there is a dearth of information. For most substances the only available data indicate a NOAEL or LOAEL, usually from animal tests. Recently Pennington et al [2002] have proposed a promising method of using LOAEL or NOAEL data for estimating DRFs, but among toxic metals their only result so far is for Hg and only for an endpoint for which no monetary valuation is available.

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

For the carcinogenic metals, As, Cd, Cr (in oxidation state 6) and Ni, the CRFs given by EPA are stated as unit risk, shown here in the second line of Table V.2.2; they are the probability, per $\mu g/m^3$ of ambient concentration, of getting a cancer due to a lifetime exposure, taken as 70 yr. With our definition of the CRF as impact for a 1 yr exposure, the slope s_{CR} of the CRF is the unit risk divided by 70. The fifth row shows the cancers per kg of emitted pollutant, based on the inhalation dose only. At the present time the evidence for cancers due to the ingestion of Cd, Cr and Ni is not sufficiently convincing for EPA to indicate a DRF. For the social cost of cancers we take the value of about 2 M€ per cancer used by the ExternE project series; it is an average over fatal and nonfatal cancers and takes into account the shortening of life expectancy. If other endpoints were included, the cost would be higher, but we do not know how much.

For As ingestion is considered carcinogenic, with slope factor 1.5 per mg/(kg.day). Since the slope factor indicates the lifetime risk due to ingesting the same dose every day for 70 yr, we need to divide by 70×365 days and the average weight of 55 kg/pers to obtain the DRF in our units. Multiplying the resulting slope s_{DR} of the DRF by the ingestion dose in Fig.V.2.2, we find 3.05E-04 cancers per kg of As due to ingestion, much more than due to inhalation although not in proportion to the dose ratio; ingestion seems to cause less cancers per dose than inhalation. The 3.05E-04 cancers per kg of As due to ingestion may be a serious overestimate because it assumes the same toxicity for organic and for inorganic As. At the present time EPA and the International Agency for Research on Cancer do not provide any information on the carcinogenicity of organic As. Most of the ingestion dose is organic, with the exception of drinking water which is inorganic. Taking only the dose from drinking water, the cancers from ingestion are 0.17E-05 cancers per kg of As due to ingestion, as shown in Table V.2.2. Even that is an overestimate if some of the As is removed by water treatment.

Table V.2.2. CRFs, DRFs and impacts, per kg emitted, for the carcinogenic metals. Unit risk and slope factor from the IRIS database of EPA http://www.epa.gov/iris.

	As	**Cd**	**Cr**	**Ni**
Inhalation				
unit risk [cancers/(pers·70yr·µg/m3)]	4.30E-03	1.80E-03	1.20E-02 [a]	2.40E-04
s_{CR} [cancers/(pers·yr·kg/m3)]	6.14E+04	2.57E+04	1.71E+05 [a]	3.43E+03
Cancers/kg, inhalation, UWM	**2.32E-05**	**9.73E-06**	**0.84E-05** [b]	**1.30E-06**
Ingestion				
slope factor [cancers/(mg/(kg_{body}·day))]	1.50E+00			
s_{DR} [cancers/kg]	1.07E+00			
Cancers/kg, ingestion	**1.7E-05**			
Total cancers/kg	**4.0E-05**	**9.73E-06**	**0.84E-05** [b]	**1.30E-06**
Cost/kg [€/kg] at 2 M€/cancer	**80**	**19**	**17** [b]	**2.6**

[a] for Cr-VI
[b] assuming that only 13% of the Cr emitted by power plants is Cr-VI

Finally we calculate the impact and damage cost of IQ decrement due to Pb, a cost that can be quantified with present knowledge and probably the dominant part of the total damage cost of Pb. The DRF is quite well determined, thanks to a meta-analysis by Schwartz [1994] who found a decrement of 0.026 IQ points for a 1 µg/l increase of Pb in blood, a relation that appears to be linear without threshold. More recently a study designed to identify effects at the lowest doses found an even larger effect, 0.055 IQ points per 1 µg/l, without any threshold [Lanphear et al 2000]. Here we continue to use 0.026 IQ points per 1 µg/l, being based on a meta-analysis rather than a single study.

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

We also need to consider the time window during which an exposure causes damage. The sensitivity of the brain to Pb is greatest during the first two years of life, although the precise time distribution of the damage is not known. However, this does not matter since the result of Schwartz expresses the total impact in a population due to a constant exposure.

Furthermore, the half life of Pb in blood and other soft tissues is relatively short, about 28-36 days (although much longer in bones) [WHO 1995]. Thus, for the purpose of damage calculations, one can equally well assume that the damage is incurred during a one year exposure by infants between the ages of zero and one only, or during a three year exposure between the ages of zero and three. To see that the effect is the same, note that the percentage of the population between zero and three is essentially three times the percentage between zero and one, the latter being 1.1% in the EU. If the sensitive period is only one year, the loss due to a one year exposure is 1.3 IQ points/(μg/m^3) × 1.1% of population of EU. If the sensitive period is three years, the affected cohort is essentially three times as large but the damage rate three times smaller, so the loss due to a one year exposure is (1.3 IQ points/(μg/m^3))/3 × (3 × 1.1% of population of EU), essentially the same. To express the CRF slope in a form consistent with this paper, i.e. relative to the entire population, we therefore multiply the 1.3 IQ points/(μg/m^3) by the fraction of the population that is affected (1.1% per year), to obtain

s_{CR} = 1.43E-2 IQ points/(pers·yr·(μg/m^3)). (V.1)

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

Multiplying the inhalation dose by a factor 20 for automotive emissions in cities, and combining it with the CRF of Eq.V.1 we find a loss of 0.268 IQ points per kg of Pb. For the cost associated with the loss of an IQ point we take 3000 €/IQ point, based on numbers cited by Lutter [2000]. Thus we find a damage cost per kg of Pb emitted in Europe of

3000 €/IQ point × 0.268 IQ points/ kg = 804 €/kg, for automotive emissions in cities, (V.2a)
based on the relation between blood level and concentration in air;

for power plants we do not multiply by the factor 20 and obtain

3000 €/IQ point × 0.013 IQ points/ kg = 40 €/kg, for power plant emissions, (V.2b)
based on the relation between blood level and concentration in air.

The second option is a relation between blood level Pb and ingestion dose, published by WHO [1995]. Surprisingly the blood level per ingested quantity is higher at low doses, perhaps because of increased excretion at higher dose or storage in bones. Here we use the level found at the lower dose, 72 μg/l for infants who ingest 17 μg/day, or 4.2 μg/l per ingested μg/day. Together with the above mentioned 0.026 IQ points per 1 μg/l increase of blood Pb this implies a loss of 0.026 IQ points × 4.2 (μg/l)/(μg/day) × (1 yr/365 days) = 3.02E-04 IQpoints/(μg/yr) per child. As in the argument leading to Eq.33 we multiply this number by 1.1%, the fraction of the total population below 1 yr of age and sensitive to Pb, to obtain a DRF slope of

s_{DR} = 3.32E+03 IQpoints/(pers.kg). (V.3)

Again the duration of the sensitive period during infancy does not matter. Multiplied by the collective ingestion dose of 1.64E-04 kg per kg emitted (with 100 yr cutoff), this yields an impact of 0.54 IQ points per emitted kg, and at 3000 €/IQ point the cost is

3000 €/IQ point × 3.32E+03 IQpoints/(pers.kg) × 1.64E-04 kg/kg = 1633 €/kg, (V.4)
based on the relation between blood level and ingestion dose.

GREENSENSE [2003] offers another estimate of the unit cost, 8600 €/IQ point, based on lost earnings; with that the damage cost of Eq.V.4 would be 4680 €/kg. The result of Eq.V.4 is almost the same for automotive and for power plant emissions, regardless of emission site, because the inhalation dose is such a small part of the total dose.

We have more confidence in the relation of blood level with ingestion than with inhalation. The relation with inhalation appears less reliable, because the inhalation/ingestion ratio is likely to be quite variable with site, and over time as well, especially with the phasing out of leaded gasoline. In view of this range of possible estimates we tentatively recommend a value of

1600 €/kg, for Pb emissions, with negligible differences between emission sources, (V.5)
based on the relation between blood level and ingestion dose.

However the uncertainties are very large.

One motivation for this calculation is the fact that even so-called unleaded gasoline can contain Pb: the regulatory limit for unleaded gasoline in the EU after 2000 is 5mg/l [EC 1998]. At this level, and with 1633 €/kg, the associated damage cost would be 0.008 €/l, small but not negligible compared to the price of gasoline.

V.3 The WATSON model

Like the UWM above, the integrated WATer and SOil environmental fate, exposure and impact assessment model of Noxious substances (WATSON) for Europe can be considered an extension of the software tool EcoSense proposed within the ExternE project [EC 1999] (see Figure V.3.1).

In order to allow for a bottom-up impact assessment approach that is in agreement with the impact pathway approach of ExternE, the media soil and water need also to be modeled in a rather spatially resolved way for the whole of Europe. Different from air, however, water and especially soils show highly variable properties so that there is quite substantial literature on the most appropriate spatial and also temporal resolution at which these media would best be modeled [e.g. Addiscott 1998][Becker 1995][Hoosbeek; Bryant 1992][Kirkby et al. 1996][Blöschl 1996]. Models that cover larger areas than just a catchment with a fair degree of spatial resolution usually operate on a grid and most often cover the whole globe as global (atmospheric) circulation models. However, their focus is on the water balance. Although the modeling based on lumped parameters at larger scales is seen very critically [Addiscott 1993][Becker 1995] the model to be developed also needs to be acceptable in terms of computing time and data storage needs as it is meant to be a decision-support tool rather than serving research purposes.

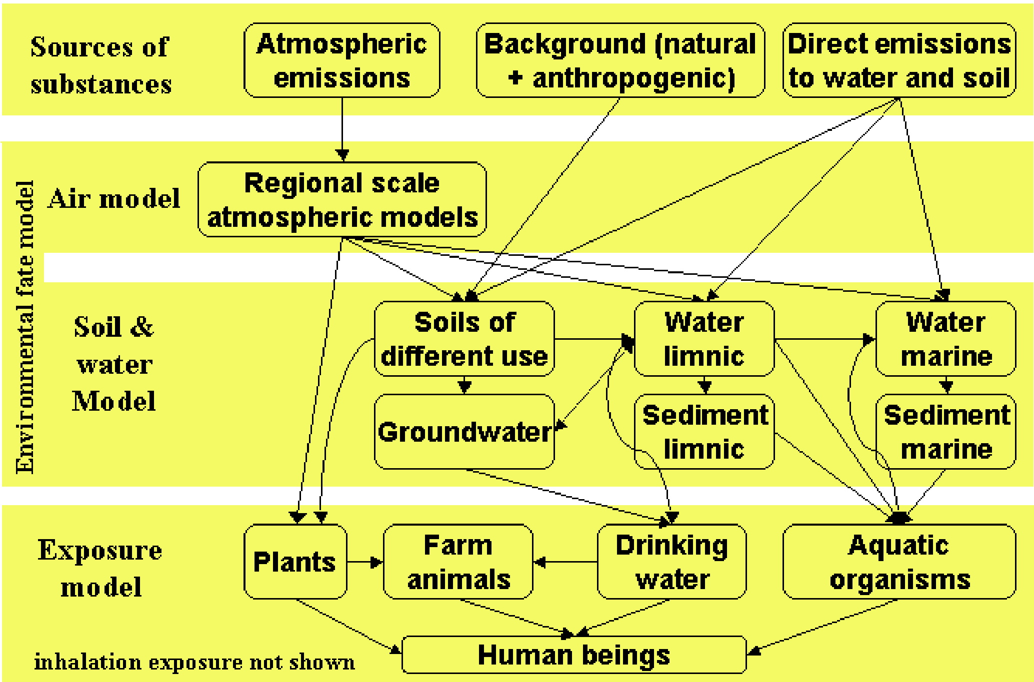


Fig.V.3.1: Conceptual structure of the multimedia model and the exposure assessment of the WATSON model (arrows denote a substance's environmental pathway; exposure via inhalation is not indicated)

As a consequence, the multimedia modeling approach has been followed here [Mackay 2001] which is well suited to quantitatively assess average concentrations at the regional scale resulting from highly dispersed and diffused sources [SETAC 1995]. It is based on a mass balance that is formulated as a set of linear first order ordinary differential equations. With the help of these models, usually the steady-state solution is computed which assesses the situation when no mass change in any compartment modeled occurs due to continuous release of a substance over longer time periods. The time period until such a steady-state is reached actually depends on the nature of the substance (i.e. the related inputs and outputs with respect to the scope of the model). Therefore, WATSON offers the opportunity not only to calculate a substance's environmental concentration in water and soil as a steady-state concentration (which may serve as an indicator for sustainability if compared to a societal target value) but also dynamically with variable time steps. In addition, the time to reach a specified percentage of the steady-state concentration can be computed in order to get an impression of what time scales we have to deal with under a certain emission scenario. Different from many existing multi-media models WATSON offers the option to switch particular processes on and off according to the nature of the substance rather than setting parameters to unreasonable values [e.g. for vapor pressure of metals other than Hg in Guinée et al. 1996] since for different substances different processes are of varying importance. The processes that are covered by WATSON can be divided into different types (Table V.3.1). The processes considered are also given.

Table V.3.1 Process types and related processes in WATSON

Process type	Processes
Transformation	Degradation; decay
Exchange	
- inter-regional	River discharge; circulation of large lakes[a]
- intra-regional	Matrix leaching and *preferential flow;* soil erosion and Hortonian overland flow; sedimentation, resuspension and *sediment burial*; ice melt of glaciers; diffusive exchange between water and sediments
Direct and diffuse input	Dry and wet atmospheric deposition; direct releases into water and soil

[a] if a lake is fully contained in a region it is already assumed to be fully mixed or homogeneous as part of a freshwater compartment according to multimedia modelling practice.

One drawback for coupling an air quality model to a multimedia (soil and water) model could be that it is not fully integrated. This means that the assumed/expected multiple intermedia exchanges between air on the one hand and soil and water on the other of for instance the so called multimedia organic pollutants may not be warranted. For the bulk of substances which are not true 'multi-hop pollutants' [Klepper; den Hollander 1999], however, the intermedia exchange (or feedback) is assessed to be small for the bulk of organic chemicals [Margni et al. submitted]. Heavy metals can principally re-enter the atmosphere via volatilisation and resuspension when attached to particles. Apart from mercury, heavy metals do not have a significant vapor pressure so that volatilisation can be neglected. Suzuki et al. [Suzuki et al. 2000] investigated the influence of wind erosion on the fate of rather persistent organic chemicals with the help of a (fully integrated) multimedia model. In a sensitivity analysis, they found that this process is negligible. Therefore, it is assumed here that also for (persistent) heavy metals this process can be neglected.

The feedback of organic substances can be taken into account when defining the air quality model's exchange rates with the respective ground surface for the particular 'multi-feedback' substance (see table V.3.2 for examples of feedback fractions).

Table V.3.2 Feedback fractions of selected substances [Margni 2002]

Substance	Feedback fraction [%]
Benzene to air / to water	2 / 1
2,3,7,8-TCDD	0.2
Benzo[a]pyrene	$9.1*10^{-4}$

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

V.3.1 Environmental fate modelling

As already outlined above, the environmental fate model consists of an existing single-medium air quality model (the Windrose Trajectory Model WTM) linked to a water and soil multimedia type of model (‘air model’ and ‘water and soil model’ bars in Figure V.3.1). The multimedia soil and water environmental fate model divides Europe into about 3400 so called base regions (see Figure V.3.2) according to the HYDRO1k GIS dataset for basins [USGS 1996] (for comparison: the air quality model WTM is based on the EMEP 50 grid with 6600 terrestrial grid cells in Europe). This dataset was derived from a digital elevation model on a 1 km2 raster. Although it contains some deviations from the real water pathways over the land surface it allows a complete division of Europe into drainage basins. Deviations that had been detected and considered severe by comparing to the European rivers and catchments database [ERICA Version 1998, EEA Data Service 1998] as well as to the Britannica Atlas [Cleveland et al. 1984] were corrected. Each drainage basin generically consists of different compartments, i.e. soils of different land use (i.e., pastures, arable land, non-vegetated areas (e.g. rocks, open cast mining), semi-natural ecosystems (e.g. forests, heathlands), built-up areas, glaciers) and surface water bodies with corresponding sediments. At present no seawater compartment and corresponding sediment are included. Due to marine currents and migrating animals there would be a need to model the entire oceanic system on Earth for long-lived substances which in turn are the substances of highest concern. As a consequence the modeling framework is as yet not capable of estimating the exposure due to marine fish consumption which to rather high degrees contribute to exposure to e.g. methyl-mercury or dioxins [e.g. EPA 1998b][Buckley-Golder 1999][DG Health 2000].

V.3.2 Exposure and impact modelling

The predicted environmental concentrations from the environmental fate module are used to assess the exposure to living organisms and finally to humans (Figure V.3.1 ‘exposure model’ bar). There are basically three routes of exposure which may lead to an impact: inhalation, ingestion and/or dermal contact. For inhalation, a combined exposure and impact assessment approach is followed by using exposure-response functions as they have been widely applied in the series of ExternE projects [Friedrich; Bickel 2001][EC 1999].

Besides direct exposure via inhalation (not indicated in Figure V.3.1) the main indirect exposure route is ingestion of food and drinking water; dermal exposure as the third main route of exposure was left out in this investigation as this route of exposure to environmental pollutants is of much less concern compared to occupational exposure and exposure via cosmetic products. Modeling drinking water exposure for all European residents is a task that nobody has until now addressed following a detailed site-dependent bottom-up approach that aims at giving best estimates rather than those based on conservative (reasonable) worst-case scenarios. This is because it is groundwater that constitutes a major part of the drinking water resources [EEA 1999]. Even at smaller scales one fails to model mass transfers in groundwater aquifers due to lack of information [e.g. Eggleston; Rojstaczer 2000]. It also appears that groundwater contamination due to heavy metals for instance is a very localized problem and is in the case of heavy metals confined to areas with former or present mining activities [EEA 1995]. Due to the lack of contamination as well as aquifer information a modeling effort would at present result in rather unreliable concentration estimates. Thus, exposure via drinking water is for the moment not included in the modeling framework.

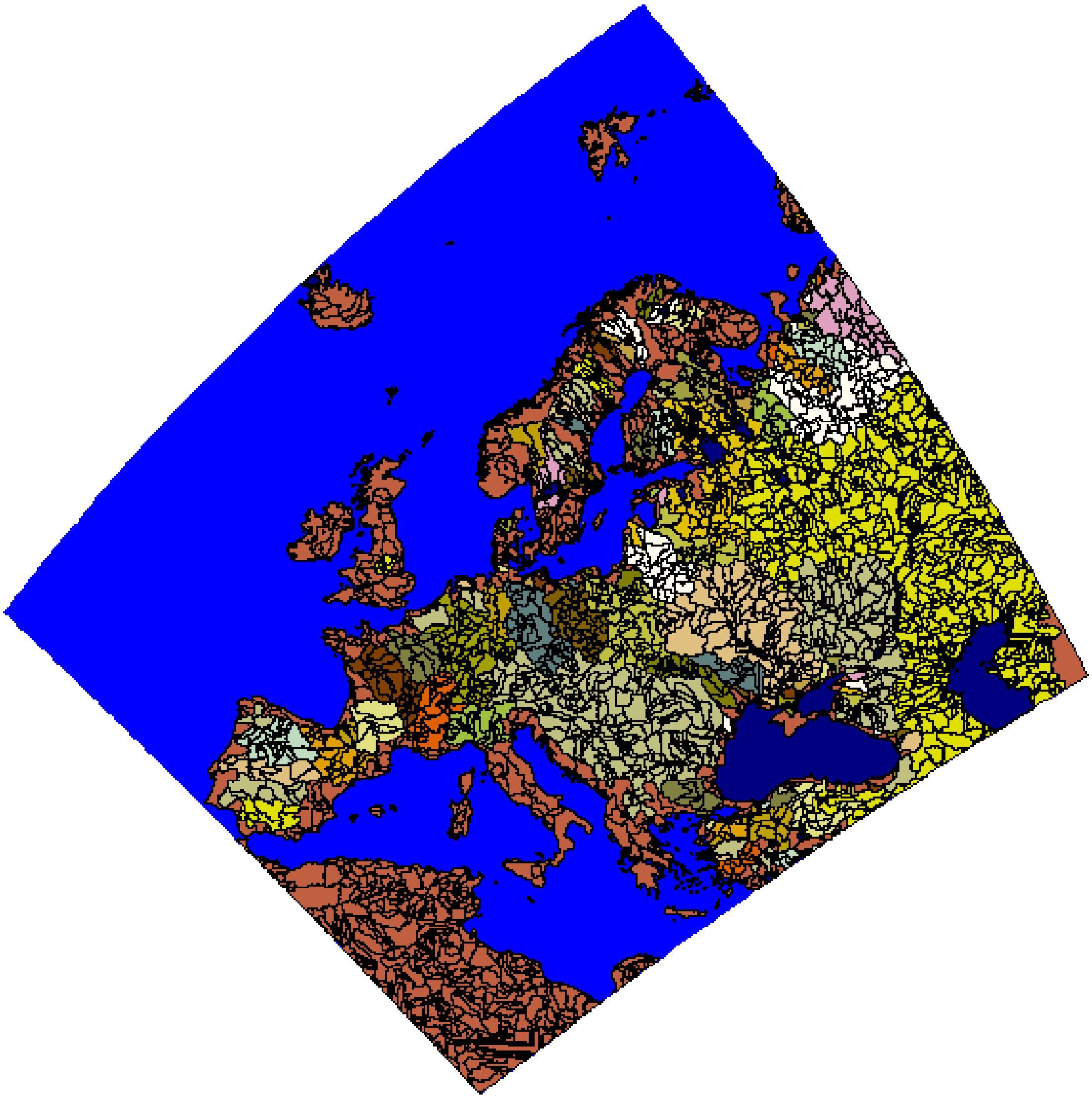


Fig.V.3.2: Geographical scope of WATSON-Europe corresponding to the receptor area of the EcoSense model. Catchments consisting of more than one region are coloured [derived on the basis of USGS 1996]

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

Since in the present study we focus on heavy metals, the exposure assessment of EPA [1998a] has been followed similar to UWM. A further restriction is that not all food ingestion related exposure pathways are included in WATSON at present. Especially exposure via fish is not considered. This is because most fish consumed in Europe stems from sea catches [ECETOC

1994]. As was argued above, however, modelling the marine environment almost inevitably brings about the necessity to extend the geographical scope of the model to the whole globe. Thus, fish consumption is as yet not included. This is also the case for pork although it is the dominating meat type consumed in Europe [ECETOC 1994]. Different from cattle, pigs are usually fed purchased feed to a large extent which is a mixture of different base feed components that are grown worldwide. Allocating the different components to production inside and outside of Europe has not yet been done which is why exposure via pork is not considered here. Although the exposure assessment due to ingestion is not exhaustive, exposure via staple food products are to a large degree considered (i.e., wheat, barley, rye, potato, beef, cow milk and products).

Different from inhalation and exposure via drinking water, the exposure via food does not only need to take into account the environmental concentration and the transfer into plants and/or animals but also the trade of the food that contains a substance which causes an adverse effect. As NewExt is a European project only the trade within Europe shall be considered. For this it is assumed that the food items are equally distributed over the whole European/Asian receptor area of WATSON (see Figure V.3.2) weighted by the stocks or the produced amounts of livestock and crops, respectively. These are taken from the data already implemented in EcoSense (i.e. wheat, barley, sugar beet, potatoes, sunflowers, tobacco, rye, oats). Missing data on beef, porc and milk production are taken from FAO statistical database [FAOSTAT 2002]. This approach of course is a generalization of the real path of food products or, on the other hand, of the actual exposure scenario. It is very different from typical risk assessment frameworks where the conservative ‘subsistence farmer exposure’ scenario is often used [EC 1996]. Allowing for trade rather is in line with Pennington [Pennington 2001] who introduced a ‘production-based’ approach where a so called intake fraction [e.g. Bennett et al. 2002] assesses the portion of an emission that a population will be finally exposed to. The intake fraction is, thus, a good measure to base exposure response functions on in order to get representative impact estimates.

For the discussion of dose-response or concentration-response data please refer to section V.2.

V.3.3 Scenario calculations with respect to exposure and results

Based on the emission inventory by [UBA; TNO 1997] and [EMEP 2002] where available, scenario calculations for the whole of Europe were performed for the heavy metals As, Cd, Pb, Ni, and Cr. The results were normalized to one kilogram of the respective pollutant emitted. The emission scenarios as well as the correction for local inhalation exposures have been adopted from [GREENSENSE, 2003].

Table V.3.2 gives the impacts and damage costs due to inhalation of the respective trace element emitted. In the case of chromium an adjustment to only consider chromium in the valence state of +VI was necessary since it is only in this valence state hazardous. Chromium released into the environment from combustion processes and ore processing industries is present mainly as chromium(III) oxide (Cr_2O_3) which is even essential to humans and animals [ATSDR 2000]. Chromium VI usually is formed by heating chromium(III) oxide in the presence of soda and lime at temperatures of 1100-1200°C under oxic conditions [Holleman; Wiberg 1985]. Due to primary measures taken for reducing the formation of NO_x it is assumed here that conditions in the furnace of oil and coal fired power plants which contribute the largest anthropogenic sources of particle bound chromium [UBA; TNO 1997] are not in favor of converting the dominating trivalent into hexavalent chromium. However, some hexavalent chromium could be detected in fly ash from coal-fired power plants ([EPA 1998b] and [Stern et al. 1984] quoted in [ATSDR 2000]. There is little information about chromium speciation in emissions from combustion processes. From 11 test sites it was found that the average chromium VI is 11 and 18 percent for coal-fired and oil-fired utilities, respectively [EPA 1998b]. A comparable value of 13 percent was found for shipyard welding[Mener et al. 2001]. These figures are in line with the statements by [EPA 1998c][ATSDR 2000] in that chromium emitted to air is predominantly in the +III valence state and if emitted in the hexavalent state non-energy related processes are mostly responsible.

The cancers per kg of pollutant emitted due to inhalation exposure tend to be higher by a factor of 2 to 3 than those assessed by the UWM (see Table V.2.2; note: ingestion exposure of arsenic is not included here; for reasoning see text below).

Table V.3.2. CRFs and impacts, per kg emitted, for the carcinogenic metals for inhalation exposure. (slope factor and valuation taken from table V.2.2)

	As	**Cd**	**Cr**	**Ni**
Inhalation				
s_{CR} [cancers/(pers·yr·kg/m3)]	6.14E+04	2.57E+04	1.71E+05 [a]	3.43E+03
Accumulated inhalation exposure [pers·yr·kg/m3]	1.08E-03	5.94E-04	2.89E-04 – 4.72E-04 [a]	7.02E-03
Cancers/kg, inhalation, EcoSense	**5.56E-05**	**2.93E-05**	**2.18E-05 – 3.57E-05** [b]	**2.50E-06**
Cost/kg [€/kg] at 2 M€/cancer	**111**	**59**	**44 – 71** [b]	**5.0**

[a] for Cr-VI
[b] assuming that only between 11 (average coal) and 18 percent (average oil) of the Cr emitted by power plants is Cr-VI

For indirect exposure via ingestion, so far cadmium and lead can be modeled by WATSON. This is because modeling As and Cr implies the need to consider different species. For instance, redox conditions vary in time so that As becomes more and Cr less mobile under reducing conditions [US-EPA; DOE 1999a][Jain; Ali 2000]. Changing redox conditions are mostly due to varying soil moisture contents or water bodies becoming partly unoxic over the course of the year. Redox reactions are slow processes. However, equilibria are usually reached within a couple of months [Smedley; Kinniburgh 2003] so that at least seasonal variations would be needed to be taken into account. Additionally As in food occurs as organic compounds which do not pose a threat to human health [Chaney; Ryan 1994][Harrison 2001]. In fact, the IRIS database at EPA provides a dose response function for oral uptake only for inorganic forms of arsenic. As regards the drinking water dose response function there is little evidence to suggest that atmospheric As poses a real health threat for drinking-water sources [Smedley; Kinniburgh 2002] also given the highly variable natural background. Because of this and as stated above, WATSON has not yet been developed to include drinking water exposure which is why As is not included.

Due to lack of reliable emission data only the emissions for the year 1990 are evaluated. In order to attribute the exposure to emissions of that year WATSON calculates a pulse emission that lasts one year. The total exposure aggregated over different time horizons as well as at steady-state are given in Table V.3.3. Initial background concentrations need to be set to zero in order to account for the amount of substances only released during the pulse emission.

Table V.3.3: Intake fractions of a pulse emission based on 1990 air emissions of cadmium and lead for different time horizons and different exposure routes according to exposure framework given by [EPA 1998a]

Exposure horizon	Intake fraction	Share of intake fraction due to exposure via ...						
		milk	beef	wheat	barley	rye	potato	air[a]
Cadmium								
1 a	0.0006%	0.015%	0.013%	22.9%	0.31%	3.4%	2.0%	71.3%
10 a	0.0022%	0.040%	0.036%	63.8%	0.86%	9.4%	5.7%	20.1%
30 a	0.0055%	0.046%	0.041%	73.5%	0.99%	10.9%	6.6%	7.97%
100 a	0.0157%	0.047%	0.042%	77.4%	1.04%	11.7%	7.0%	2.79%
steady-state	0.0796%	0.045%	0.039%	79.3%	1.04%	11.8%	7.2%	0.55%
Lead								
1 a	0.0006%	0.08%	0.00%	3.0%	0.04%	0.2%	0.2%	96.4%
10 a	0.0008%	0.58%	0.04%	22.9%	0.31%	1.8%	1.7%	72.8%
30 a	0.0012%	1.12%	0.07%	44.3%	0.61%	3.4%	3.2%	47.3%
100 a	0.0026%	1.64%	0.10%	66.0%	0.90%	5.1%	4.7%	21.6%
steady-state	0.0467%	1.91%	0.12%	85.0%	1.11%	5.6%	5.1%	1.22%

[a] exposure due to inhalation only occurs in the first year

Due to the retention of the heavy metals in soils the intake fraction increases with longer time frames over which the annual intakes are integrated (Table V.3.3). It appears that the intake fraction of cadmium is larger than that of lead especially in the short to intermediate term. This can be attributed to the lower mobility of lead in the environment [Kabata-Pendias; Pendias 1992] that is also reflected in the food chain transfer values [EPA 1998a] and explains why lead converges more slowly to the final intake fraction. This is also supported by the observation that cadmium is fairly readily taken up by plant root cells [Welch; Norvell 1999]. For both heavy metals, inhalation exposure dominates the total exposure in the first year making up between 71 and 96 % of the first year's intake fraction for cadmium and lead, respectively. Whereas the contribution of the different foodstuffs for cadmium exposure reaches a constant composition within a few decades this composition converges over the course of time more slowly for lead. For both metals, exposure due to ingestion of cereals dominate by far (about 90 %) followed by potatoes.

Most notable is the observation that after an elapsed time horizon of 100 years only a small fraction of the potential total intake fraction (at steady-state) has reached the human population: 20 and 6 % for Cd and Pb, respectively. This means that most of the damages of very persistent pollutants only occur at some time in the far future giving raise to concerns with respect to inter-generational justice/equity in the context of sustainable development. Additionally, the exposure assessment is not complete so that the absolute value of the intake fraction is underestimated. On the other hand, the dynamics are not expected to be highly influenced by this incomplete picture as many of the foodstuffs that people consume at large quantities are covered.

Cadmium is known to appreciably accumulate in rice and tobacco [Chaney et al. 1999] which are not included in the ingestion and inhalation exposure assessment presented above. However, rice production in Europe is fairly limited [FAOSTAT 2003] so that the contribution of European emissions that could be held responsible for elevated Cd concentrations in rice grains are considered negligible. In the case of smoking tobacco it is argued here that people are aware of the fact that smoking poses a threat to their life so that also their additional cadmium inhalation exposure is considered internalised. In terms of indoor air quality the issue of indirect smoking should at the same time be borne in mind. It is debatable whether the indirectly exposed people voluntarily accept the higher exposure situation so that their exposure again could be considered internalised.

Comparing the WATSON results to those yielded by the UWM is not readily done as the routes of exposure considered differ. Also the approaches taken with regard to emissions (pulse vs. continuous) hinders a direct comparison. With both models, however, exposure via cereals seem to dominate the exposure via ingestion whereas (short-term) inhalation exposure only contributes to a few percent to the overall intake fraction.

It can be concluded that when ingestion exposure especially of persistent pollutants is considerable very long time horizons need to be taken into account during the impact and/or damage assessment.

For a more detailed discussion of the methodology employed the reader is referred to section V.3.4.

V.3.4 Scenario calculations with respect to concentrations in soil and water and related results

In order to compute concentrations in soil and water a comprehensive set of information on emissions to air and directly to these media as well as existing natural and/or anthropogenic background concentrations would be required. For the year 1990, a European wide inventory of air emissions exists [UBA; TNO 1997] which can be supported by [EMEP 2002]. A European wide inventory of direct emissions to water and soil is planned by the European Union but is unfortunately not yet available. National inventories on direct emissions of toxic substances to single media already exist in different countries. However, resource constraints did not allow a compilation of such data. In order to consider the natural background constant concentrations will be assumed (see below).

The concentration data will be presented for the steady-state situation as well as for a time horizon of 1000 years. In contrast to the 100 year cut-off employed above a value of 1000 years is chosen since human activities that released metals into air have been going on for decades if not centuries (e.g. metal processing). In the absence of comprehensive historic emission records and in order to take these former emissions into account a value larger than the 100 years used above is chosen. However, one may argue that former emission strengths had been smaller than those reported for the year 1990 so that assuming these emissions to last for 1000 years definitively overestimates the anthropogenic background encountered today.

Table V.3.4 shows the maximum concentration at steady-state as well as after 1000 years with respect to environmental media. Some results are given as ranges because WATSON does not take into account the naturally occurring metal concentration which to a rather high degree needs to be freed via weathering prior to more actively participating in the distribution and especially exposure processes. The model rather only considers the mobile anthropogenic fraction of the metals. The natural background of these metals varies to about two orders of magnitude in different soils also due to variation in the type of parent material [Kabata-Pendias; Pendias 1992][Reimann; de Caritat 1998][Nriagu 1978][Merian 1991]. For the time being the natural background concentration is considered by adding to the steady-state environmental media concentrations a constant background representative for the upper continental crust of 0.1 ppm and 17 ppm for cadmium and lead, respectively [Wedepohl 1995].

Table V.3.4 Maximum concentrations at steady-state and after 1000 years for air emissions in 1990 and exceedances of the standards in terms of area fractions affected for the environmental media concentrations (note: variable units; ranges indicate situation when background concentration was considered)

	Maximum concentration		Compliance check	
	at steady-state	after 1000 a	Standard	Exceedances at steady-state [fraction of area affected]
Any soil [mg/kg DW]				
Cd	1.57-1.67	0.42-0.52	0.3[a]	0.6-1.6%
Pb	2738-2755	21-38	38[a]	3.9-12.7%
Arable land [mg/kg DW]				
Cd	1.13-1.23	0.33-0.43	1[b]	0.02-0.02%
Pb	92-109	20.5-37.5	50[b]	1.8-5.4%
Surface water [mg/l]				
Cd	0.002	0.001	0.001[c]	0.01%
Pb	0.048	0.018	0.05[c]	0%

[a] source: [de Vries; Bakker 1998]
[b] source: Directive 86/278/EEC
[c] source: Directive 75/440/EEC

In Table V.3.4 it is also shown whether the estimated steady-state concentrations comply with scientific and/or regulatory standard values. Whereas for surface waters there is (almost) no exceedance of the standard expected the situation for soils is different especially concerning the scientific standards for ‘any soil’. With regard to these standards which were derived for ‘multifunctional soil use’ the predicted concentrations do not even comply after the 1000 year period. Especially for lead the maximum discrepancy between the standard and the concentration is fairly large (more than 2700 mg/kg vs. 38 mg/kg; see discussion below).
However, one needs to be cautious when using the absolute concentrations. This is because of several aspects:

- the evaluation of the absolute predicted environmental media as well as foodstuff concentrations,
- the assumption on constant environmental conditions over long time horizons, and
- localized contaminations in a regionally resolved modelling framework and under limited emission information,
- transformation processes for metals to be potentially considered.

Although steady-state mass balance models with homogeneous compartments (also known as level III models) are seen as appropriate tools to predict average regional concentrations resulting from highly dispersed and diffused sources [SETAC 1995] a thorough evaluation of WATSON still needs to be performed. Generally, the more elevated lead levels in soils correspond fairly well to its being the least mobile among the heavy metals [Kabata-Pendias; Pendias 1992]. Compared to lead, cadmium is more mobile. Elevated concentrations in top soils are usually due to anthropogenic contamination [Kabata-Pendias; Pendias 1992] and most notably due to atmospheric emissions in industrialized countries [McLaughlin; Singh 1999]. Comparing the maximum steady-state concentrations (Table V.3.4) with those observed (see Table V.3.5) reveals generally good agreement with the expectation values. There are two exceptions with respect to Pb concentrations in environmental media. 1) The lead concentrations for 'any soil' in Europe. A closer look at the highest lead levels showed that these occur on pastures and semi-natural ecosystems. Due to the permanent vegetation cover, it is assumed that soil erosion does not occur. This is in line with the crop management factor of the Universal Soil Loss Equation (USLE) being considerably smaller for these land uses than for arable land [Morgan 1999]. It appears that the areas with the highest lead concentrations across Europe are those that are not affected by erosion and at the same time have a low water volume running off the land which could entrain dissolved amounts of pollutants and receiving relatively large atmospheric deposition fluxes. It needs to be further explored whether the assumption to have no soil erosion on pastures as well as on areas with semi-natural ecosystems is valid in such a strict way. However, it shows the importance of this process for very persistent substances. 2) Lead concentrations in the bulk freshwater exceeds the median observed values in Table V.3.5. However, it is still in the total range.

Steady-state concentrations may serve as indicators in the context of sustainability assessments. However, in Table V.3.4 it was shown that the time until the steady-state concentration is reached may be well on the order of several hundreds of years. In the light of the problems related to e.g. climate change and acid rain it seems rather unrealistic to assume constant long-term environmental situations. Especially values for the pH and organic carbon content of the environmental media, for the hydrological cycle (e.g. rainfall, evaporation), and consequently for the areas under specific land uses and related agricultural production for the exposure assessment are definitively changing over time. The same applies to population figures used in the exposure assessment.

There exists a wide-spread contamination with lead and cadmium due to e.g. metal mining and processing, sewage sludge and fertilizer additives, and local depositions on roadside soils (see Table V.3.5) which is not reflected in the investigated emission scenarios. The higher depositions along streets cannot be appropriately estimated in the current model due to the regional spatial resolution. As stated above, emission inventories to soils do not as yet exist which is why sewage sludge applications are not included in the emission scenarios. With respect to existing background concentrations, however, only continuous releases rather than initial concentrations influence the final situation at steady-state. It is, thus, considered worthwhile to include another source term that accounts for releases e.g. due to weathering of anthropogenic and/or natural background stocks in the future. Local hot spots due to continuous releases on the other cannot at present be accounted for adequately.

Having seen above that soil erosion is very important for the environmental dynamics of persistent pollutants it needs to be scrutinised whether all relevant processes are covered as yet. At present, the only final removal processes of metals from the modelled system are discharge to the marine environment, sediment burial and leaching to the subsurface. For metals, no transformation reaction be it speciation (e.g. [Ure; Davidson 1995]) or inactivation due to irreversible binding (e.g. [Selim; Amacher 2001]) or precipitation of insoluble minerals (e.g. [Robarge 1999]) has been included. Considering speciation which is crucial e.g. when trying to include trace elements like arsenic and chromium is in principle possible by adding the same amount of equations per considered species in the environmental fate model as for the present single species version which in turn may lead to very large equation systems. Irreversible binding, however, is another issue. By ‘irreversible binding’ one means that the release of substances that are sorbed to ‘geo-media’ is kinetically hindered and practically impossible [Lumsdon; Evans 1995]. Irreversible binding as any binding is, however, dependent on available surfaces which is why this process is capacity-limited. The net ‘irreversible’ nature is due to sorption at specific sites that have a higher affinity for the respective metal (higher binding strengths). Especially these binding sites become less and less available with a higher degree of ‘irreversible’ binding occurring until the capacity is exhausted [Selim; Amacher 2001]. Models that are used to describe sorption processes (e.g., Langmuir and Freundlich isotherms as well as surface complexation models [Jenne 1998]) take this binding capacity into account and are as a consequence non-linear. The degree of occupancy of the specific sorption sites is also the reason why solid-water partitioning values for some metals (like copper, cadmium and lead) are dependent on the overall metal concentration [Selim; Amacher 2001][US-EPA; DOE 1999b]. This is also true for the dynamic equilibrium between a precipitate and the dissolved metal fraction. Introducing parameters that depend on the substance’s concentration into an environmental fate model would require to formulate it with non-linearities. Thus, the approach followed in this study which is suggested e.g. by [SETAC 1995] that is formulated as a set of ordinary first order linear differential equations would have to be abandoned.

Another aspect of considering the inactivation processes explicitly is that plant uptake as used in many exposure models and also by both models employed in this study [IAEA 2001][EPA 1998a] is based on a transfer factor relating the total dry soil concentration to the plant concentration. In the analysis of the total dry soil concentrations usually strong agents like nitrohydrochloric acid (‘aqua regia’) or hydrofluoric acid are used which would even release at least to some degree the ‘irreversibly’ bound and precipitated fractions of the metals irrespective of their availability under natural conditions. Thus, there is a need to also include the inactivated metals in the bulk concentration numbers. However, if the process of inactivation of metals be it due to irreversible binding or due to precipitation was to be introduced into the steady-state environmental fate model it would need to be formulated as an overall loss from the system removing the amount of metals at the same time from the bulk soil. This in turn would not allow to consider this fraction in the bulk soil concentration for plant root uptake estimates. Furthermore, for consideration of speciation and inactivation processes influences of the oxidative power (or redox conditions) of an environmental medium that varies diurnally or seasonally cannot (e.g. [Olivie-Lauquet et al. 2001][Bartlett 1999]) be dealt with if employing a climatological model that makes use of long-term annual meteorological and hydrological information. As a result, one would have to abandon the level III (steady-state)/IV (dynamic) modelling approach which would require the development of a new model in order to allow for irreversible binding and redox conditions adequately.

In the study a multimedia framework has been developed in order to assess the environmental concentrations of and exposures to toxic substances like lead and cadmium in a spatially resolved way for the whole of Europe. The calculations show the variability in time and space of the concentrations. Although not replacing a more thorough model evaluation the concentrations compare well with observed concentrations.

V.4 The Vlier-humaan (VH) model

Even though the VH model is by concept totally different from the Uniform World Model as well as from the EcoSense-WATSON model, it can provide certain checks. In the following a more detailed model comparison between VH and UWM will be presented because parameter values can be more easily adopted for scenario analyses from a single-zone model (UWM) than from a rather highly spatially resolved model (EcoSense-WATSON).

Table V.3.5 Measured cadmium and lead concentrations in environmental media and cereals at the global

source	Cadmium concentration [Reimann; de Caritat 1998] [a]	Cadmium concentration [Kabata-Pendias; Pendias 1992] [b]	Lead concentration [Reimann; de Caritat 1998] [a]	Lead concentration [Kabata-Pendias; Pendias 1992] [b]
Any soil [mg/kg DW]				
total range	<0.01-40.9	0.01-4	3-16 338	1.5-286
expectation value (range)	0.117-0.7	0.06-1.1	7.45-40	7.9-84
maximum value due to local contamination				
- metal processing	n/a	1 781	n/a	18 500
- roadside	n/a	10	n/a	3 916
Agricultural soil [mg/kg DW]				
total range	<0.01-3.8	0.037-0.908 [c]	3-192	1.5-888 [d]
expectation value (range)	0.117-0.3	n/a	7.45-14	10-247 [d]
maximum value due to local contamination	n/a	167	n/a	3 916
Surface water [mg/l]				
total range	$<2\ 10^{-6}$-$9.6\ 10^{-3}$	n/a	$<10^{-5}$-0.58	n/a
expectation value (range)	$<2\ 10^{-5}$-$2.9\ 10^{-4}$	n/a	$2.1\ 10^{-4}$-$3.4\ 10^{-3}$	n/a

[a] expectation values: median; soils: only top soils and particles smaller than 2 mm considered; surface freshwater: unfiltered
[b] expectation values: arithmetic mean
[c] data quoted in [Traina 1999] for soils in the USA
[d] data quoted in [Nriagu 1978] for soils in Canada

We have been able to check the ratios of ingestion to inhalation for Cr, Ni and Pb; for Ni it is very consistent between VH and UWM and for Pb reasonably consistent, given the uncertainties. For Cr the agreement is less good, possibly because of differences in the detailed chemical form of the Cr. Furthermore, we have compared some of the key parameters, such as bioconcentration factors.

To verify the consistency of the ingestion/inhalation ratios we have set the most critical parameters of VH to the values in the UWM; this has been done in several steps to identify where the differences arise.

Bio-concentration factors and soil-water distribution constants can easily be adapted. The food consumption and the fraction of polluted food therein is somewhat different in both models. We set the amounts of food consumed per person and per year the same for meat, milk products and vegetables, and for water. For the purpose of the VH model fish was of no concern, and is left out as a source of pollution. Given the small quantity of fish in the overall diet this is not a big problem, except perhaps for specific pollutants (e.g mercury). In the framework of VH and its main use, the ingestion of contaminants due to cereals is only included to calculate background doses. In our attempt to calculate incremental intake due to metal emissions, the background is set to zero, hence cereal consumption is missed. Given the high annual consumption rate of cereals in UWM, this might influence the result of the VH calculation with the UWM parameters to an unknown extent.

In summary, for nickel, we started with an emission of 1000 kg/year, calculated the inhalation dose for the most critically exposed person, and used the VH model with the following model parameters set to the values in the UWM model:

- Soil/water distribution constant (0.065 m_{wat}^3/kg_{soil})
- Annual consumption rates per person of milk & milk products (250 kg/pers.yr), meat (100 kg/pers.yr), vegetables and fruit, and water (600 l/(pers.year))
- BCF and BTF for uptake to plants and uptake to cattle
- We assumed a 100% of polluted fraction, and an agricultural land use type, which is the most critical path.

We have made calculations for the following scenarios:

- VH with the original parameters **VH-original**
- VH with the UWM parameters **VH+UWM**
- VH with the UWM parameters, but the BCF set to the Flemish values **VH+UWM-BCF**
- VH with the UWM parameters, but the soil water distribution coefficient (Kd) set to the Flemish values, for different pH values **VH+UWM-Kd pH =X**

Table V.4.1. Comparison VH - UWM for Ni

Ni (kg/(pers.yr))	**VH-original**	**VH+UWM**	**VH+UWM-BCF**	**VH+UWM-Kd pH = 4**	**VH+UWM-Kd pH = 6**	**VH+UWM-Kd pH = 8**
Soil intake	4.9E-09	-	-	-	-	-
Vegetables	1.9E-07	1.5E-08	8.0E-08	1.5E-08	1.5E-08	1.5E-08
Meat	9.6E-09	1.9E-07	3.4E-07	1.9E-07	1.9E-07	1.8E-07
Milk	1.6E-09	8.1E-08	1.4E-07	7.8E-08	7.7E-08	7.7E-08
Water	4.2E-08	3.4E-07	3.4E-07	1.1E-07	3.4E-08	1.1E-08
Dose from ingestion	**2.5E-07**	**6.3E-07**	**9.1E-07**	**3.9E-07**	**3.1E-07**	**2.9E-07**
Dose from inhalation	**3.7E-09**	**3.7E-09**	**3.7E-09**	**3.7E-09**	**3.7E-09**	**3.7E-09**
Total dose/inhalation	68	173	248	107	86	79

Compared to the ratio found in UWM of 71.3 there seems to be a good match between the local model and the regional model in terms of division of dose between ingestion and inhalation. To see whether this outcome is a coincidence, we looked at lead and chromium.

We did the same exercise: setting the parameters to UWM, and then adjusted some of the parameters when it seemed that they dominated the result.

Chromium VI is very mobile in a soil, while chromium III is not. Choice of soil water coefficient can be very crucial in this case. We tested the following scenarios:

- VH with the original parameters: **VH-original**
- VH with the UWM parameters: **VH+UWM**
- VH with the UWM parameters, but the soil water distribution coefficient (Kd) set to the Flemish values at pH of 6: **VH+UWM-Kd**
- VH with the UWM parameters, but the soil water distribution coefficient (Kd) set to the Flemish values at pH of 6, and the dilution in drinking water multiplied with factor 10: **VH+UWM-Kd+ dilutionx10**
- VH with the UWM parameters, but the soil water distribution coefficient (Kd) set to the Flemish values at pH of 6, and no drinking water pathway: **VH+UWM-Kd-no water**

It seems that UWM takes surface water concentrations as a basis for drinking water concentrations and intake, whereas VH typically looks at leaching to groundwater and from there to drinking water after a certain dilution. This may be a significant difference between the models, so leaving out the drinking water route is probably better for comparison reasons.

Table V.4.2. Comparison VH - UWM for Cr.

Cr kg/(pers.yr)	**VH-original**	**VH+UWM**	**VH+UWM-Kd**	**VH+UWM-Kd+ dilutionx10**	**VH+UWM-Kd-no water**
Soil intake	6.5E-08	0.0E+00	0.0E+00	0.0E+00	0.0E+00
Vegetables	9.1E-07	1.1E-07	1.1E-07	1.1E-07	1.1E-07
Meat	1.4E-06	2.0E-06	3.1E-06	1.7E-06	1.6E-06
Milk	1.7E-07	1.3E-06	2.1E-06	1.2E-06	1.1E-06
Water	6.9E-05	1.5E-05	5.7E-05	5.7E-06	0.0E+00
Dose from ingestion	**7.2E-05**	**1.9E-05**	**6.2E-05**	**8.7E-06**	**2.7E-06**
Dose from inhalation	**3.7E-09**	**3.7E-09**	**3.7E-09**	**3.7E-09**	**3.7E-09**
Total dose/inhalation	19561	5081	16902	2362	747

The total to inhalation ratio in UWM of about 60 (without the water intake) is much lower than the lowest ratio for VH. The mobility characteristics of Cr-VI vs. Cr-III should be looked at more closely.

Finally for lead we calculated ingestion to inhalation ratios in the following scenarios:

- VH with the original parameters: **VH-original**
- VH with the UWM parameters: **VH+UWM**
- VH with the UWM parameters, but no drinking water pathway: **VH+UWM-no water**

Table V.4.3. Comparison VH - UWM for Pb.

Pb kg/(pers.yr)	VH-original	VH+UWM	VH+UWM-no water
Soil intake	1.1E-07	0.0E+00	0.0E+00
Vegetables	4.7E-06	4.1E-07	4.1E-07
Meat	5.9E-08	2.4E-07	2.4E-07
Milk	1.9E-07	5.0E-07	5.0E-07
Water	4.2E-08	5.5E-07	0.0E+00
Dose from ingestion	5.1E-06	1.7E-06	1.1E-06
Dose from inhalation	**3.7E-09**	**3.7E-09**	**3.7E-09**
Total dose/inhalation	1404	464	313

Compared to the steady state ratio in UWM, which is about 100, there is agreement within the same order of magnitude.

V.5. Emissions inventory

The most complete inventory can be found in the report EPA [1998b] which provides detailed data for air emissions from gas, oil and coal fired power plants in the USA. Much of the data in this report is relevant for Europe, all the more so since a significant fraction of the coal burned in Europe is imported from the USA. In particular, for a given pollution control technology the emissions are proportional to the content of the pollutant in the fuel. Since the content of toxic metals and other pollutants in the fuel is highly variable from one source of the fuel to another, it is instructive to look at the range of values for different sources of coal in the USA. Table D-8a of EPA [1998b] provides data on the content of pollutants in coal from each coal producing state of the USA. Fig.V.5.1 shows, as gray error bars, the ratios Minimum/Mean and Maximum/Mean for bituminous coal, Mean being the average for the USA; in analogous manner the thin black error bars indicate (Minimum-StandardDeviation)/Mean and (Maximum-StandardDeviation)/Mean. The absolute values of the concentrations are shown in the labels, as ppm weight. Obviously the variability is very large, implying much uncertainty for any particular power plant if the composition of the coal has not been measured.

The average air emissions from fossil fueled power plants in the USA are shown in Table V.5.1. To provide an indication of the variability, the last column shows how the total emissions have changed between 1990 and 1994. For coal the average has not changed much, but for some pollutants (e.g. Pb) the emission has decreased, while for others (e.g Hg it has increased); presumably this is due to changes in the coal supply.

There are some uncertainties in the assumed values for conversion efficiency, an assumption that is necessary because the EPA report does not provide detailed data of emissions per kWh of electricity. However, the resulting uncertainties are small compared to uncertainties that arise in any specific application from not knowing the exact composition of the fuel.

Thus the 1990 data are representative of technologies of the early nineties in the USA. Since the pollutants in Table 1 are proportional to PM10 emissions, and since PM control at the time was quite similar in the USA and in the EU (mostly electrostatic precipitators), the data also seem appropriate for power plants in the EU during the early nineties. For newer power plants in the EU the values can be reduced in proportion to the respective PM emissions. Data for the coal fired power plants of Belgium are shown in Table V.5.2.

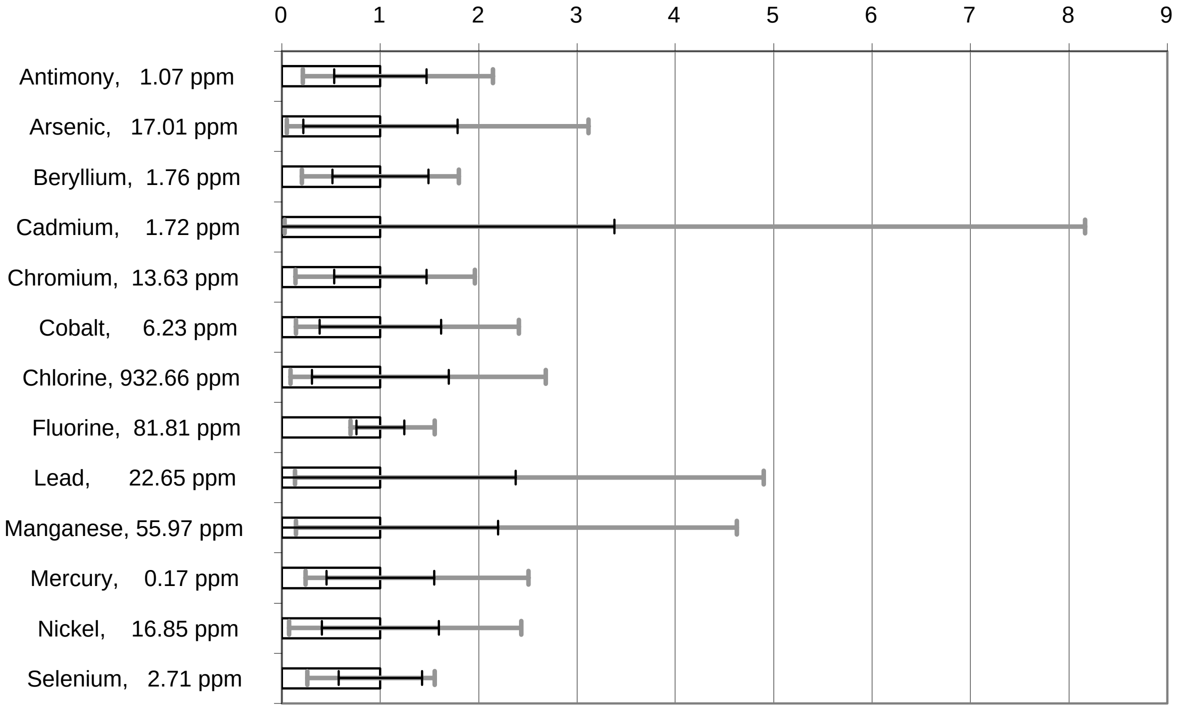


Fig.V.5.1 Concentration of pollutants in bituminous coal, for all coal producing states of the USA [EPA 1998b]. Mean concentration of each pollutant is indicated in the labels, Mean being the average for the USA; the graph is normalized by the means. Gray error bars = ratios Minimum/Mean to Maximum/Mean; black error bars = (Mean-StandardDeviation)/Mean to (Mean+StandardDeviation)/Mean.

Table V.5.1. Emission of selected pollutants from electric power plants in USA in 1990. Based on Table 3- 3 of EPA [1998b]. The last column shows ratio of total emissions in 1994 and 1990.

	mg/kWh$_e$	**ratio 94/90**
Coal-fired		
Arsenic	3.17E-02	0.92
Beryllium	3.71E-03	1.11
Cadmium	1.73E-03	0.95
Chromium	3.81E-02	0.84
Lead	3.93E-02	0.82
Manganese	8.53E-02	1.02
Mercury	2.38E-02	1.12
Hydrogenchloride	7.44E+01	0.94
Hydrogenfluoride	1.01E+01	1.18
Dioxin(TEQ)	5.05E-08	1.25
n-nitrosodimethylamine	3.04E-03	1.04
Oil-fired		
Arsenic	3.70E-02	0.70
Beryllium	3.39E-03	0.87
Cadmium	1.26E-02	0.64
Chromium	3.49E-02	0.82
Lead	7.80E-02	0.84
Manganese	6.84E-02	0.79
Mercury	1.84E-03	0.76
Nickel	2.89E+00	0.82
Hydrogenchloride	2.11E+01	0.73
Dioxin(TEQ)	1.18E-07	0.56
Natural-gas-fired		
Arsenic	4.74E-04	1.20
Nickel	6.92E-03	1.11
Formaldehyde	1.12E-01	1.10

Table V.5.2. Emissions data for coal fired power plants in Belgium, and comparison with Table V.5.1.

mg/kWh$_e$	Belgium 1990	USA 1990	Belgium 1990/USA1990	Belgium 2000
Arsenic	9.47E-03	3.17E-02	0.3	9.47E-03
Cadmium	8.61E-04	1.73E-03	0.5	8.61E-04
Chromium	3.62E-02	3.81E-02	0.9	3.62E-02
Lead	5.60E-02	3.93E-02	1.4	5.60E-02
Mercury	3.96E-02	2.38E-02	1.7	3.96E-02
Nickel				3.79E-02

EPA [1998b] has no explicit data for residues (fly ash, bottom ash, residues from flue gas desulfurization). Some indication can be found by combining the data in Fig.1 with data for the fuel input, since most of the pollutants of Fig.1 end up in the residues, by conservation of matter. However, the relative proportions in fly ash and bottom ash still have to be estimated.

The best European-wide emission inventory distinguishing between countries and sectors is given for the year 1990 [Berdowski et al 1997]. It covers all the heavy metals at stake in this report as well as some organic substances of concern. For multi-source emission assessments where emission strength, population density and land use patterns matter for the inhalation as well as for the ingestion exposure such an inventory is a prerequisite for assessing country-specific as well as European average impacts as is done by the EcoSense-WATSON modeling framework.

V.6. Conclusions: the cost per kg and per kWh

We have provided estimates for the damage cost of As, Cd, Cr, Ni and Pb, see Tables V.2.2 and V.3.2 and Eq.V.5 above. They are summarized in Table V.6.1, together with our recommendation for use in NewExt. Combining the latter values with the emissions data in Tables V.5.1 and V.5.2, g/kWh_e, one obtains the damage cost per kWh_e. This is shown in Table V.6.2. The damage costs due to these micropollutants are very small, much smaller than those due to NOx, PM10 and SO_2. This general conclusion is not affected by the very large variability or uncertainty of the emissions data: even if the emissions were an order of magnitude larger, the resulting damage cost would still be small. An analogous remark applies to the uncertainties of the damage cost estimates.

In general the damage costs yielded by the UWM and EcoSense-WATSON are within the order of two to three. Therefore, we recommend to take both estimates for sensitivity considerations. The assessment of lead and arsenic via ingestion, however, are considered especially uncertain (see section V.3).

Finally we note that the smallness of the damage cost per kWh_e does not imply that nothing should be done to reduce these emissions. A specific cost-benefit analysis is required for each technological abatement option.

Table V.6.1. Damage cost per kg of pollutant (from Tables V.2.2 and V.3.2, and Eq.V.5).

€/kg	**UWM**	**EcoSense-WATSON**	**Recommended**
Arsenic	80		80
of which inhalation	*46.4*	*111*	
Cadmium	19	59	39
Chromium	14 [a]- 23 [b]	44 [a]-71 [b]	29 [a]- 34 [b]
Lead	1633	n/a	1600
Nickel	2.6	5.0	3.8
Formaldehyde	0.12 [c]		0.12

[a] coal-fired power plants
[b] oil-fired power plants
[c] Inhalation only, CRF of ExternE [2000]

Table V.6.2. Emissions from Tables V.5.1 and V.5.2, damage cost per kg of pollutant from Table V.6.1, and damage cost per kWh_e.

	mg/kWhe	€/kg	€/kWhe
Coal-fired electric utility plants			
Arsenic	9.47E-03	80	7.58E-7
Cadmium	8.61E-04	39	3.4E-8
Chromium	3.62E-02	29	1.10E-6
Lead	5.60E-02	1600	9.14E-5
Nickel	3.79E-02	3.8	1.4E-7
Oil-fired electric utility plants			
Arsenic	3.70E-02	80	3.0E-6
Cadmium	1.26E-02	39	4.9E-7
Chromium	3.49E-02	34	1.17E-6
Lead	7.80E-02	1600	1.3E-4
Nickel	2.89E+00	3.8	1.1E-5
Natural-gas-fired electric utility plants			
Arsenic	4.74E-04	80	3.8E-8
Nickel	6.92E-03	3.8	2.6E-8
Formaldehyde	1.12E-01	0.12 [a]	1.3E-8

[a] Inhalation only, CRF of ExternE [2000]

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VI. EXTERNAL COSTS FROM MAJOR ACCIDENTS IN NON-NUCLEAR FUEL CHAINS

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TABLE OF CONTENTS

1. Introduction

In previous ExternE work, a major emphasis was placed on the quantification and valuation of impacts from beyond design basis accidents in the nuclear fuel cycle. However, other fuel chains also show a significant potential for severe accidents (e.g. oil fires or large spills, gas explosions, dam failures). The present project reviewed and substantially extended the knowledge on major accidents related to energy conversion activities. Based on the results for the individual energy chains the comparisons of selected accident indicators were carried out. Furthermore, for hydro power an approach using concepts from Probabilistic Safety Assessment (PSA) was proposed and some of its elements were demonstrated. In a second step, a methodology was developed to estimate external costs from major accidents, thus advancing comparability with the results earlier obtained for beyond design basis accidents in the nuclear fuel chain. Thus, this work enables for the first time a reasonably consistent and comprehensive assessment of externalities from major accidents in non-nuclear fuel chains.

The present investigation of accidents builds primarily on the extension of ENSAD (Energy-related Severe Accident Database), a comprehensive database on severe accidents with emphasis on the energy sector, established by the Paul Scherrer Institut (PSI) in 1998. The database allows to carry out comprehensive analyses of accident risks, which are not limited to power plants but cover full energy chains, including exploration, extraction, processing, storage, transports and waste management. The ENSAD database and the analyses have now been much extended, not only in terms of the data coverage but also the scope of data applications now including accident-related external costs. For the full coverage of the current state of the work on accidents, integrating also results from related PSI-projects, we refer to (Burgherr et al., to be published). The full report includes detailed results for individual energy chains, while the present one focuses on aggregated indicators most relevant for the comparative assessment and estimation of external costs. However, there is one exception, i.e. the present report provides a detailed account of data on oil spills (Appendix A) as these are important, unique for the oil chain, and do not enter comparisons.

Paul Scherrer Institut (PSI, Switzerland) was leading Work Package 5 (WP 5); University of Bath (UK) carried out the econometric valuation of severe accidents.

2. Approach

2.1 Database Implementation

At an early stage of the development of ENSAD it was decided that building a severe accident database from the scratch would neither be feasible nor efficient, particularly given the actual time and resource constraints. The survey of the existing sources of information, carried out at the beginning of this effort showed that:

a) Numerous sources of information exist; their availability, scope, development status and quality exhibits an enormous variation.

b) Commercial and non-commercial databases are available. They normally cover man-made accidents in a variety of sectors and in some cases also the natural disasters. Very few of the databases deal explicitly with energy-related accidents. If they do, the coverage concerns one specific energy carrier, for example offshore accidents. In most cases energy-related accidents constitute a not explicitly identified subset among other accidents.

c) None of the available individual databases has a satisfactory coverage to form alone a basis for the evaluation of severe accidents.

d) The information assembled in the available databases even if combined, would not be fully adequate for meeting the objectives of this work. It needs to be supplemented by additional sources in order to achieve reasonable completeness and quality.

As a result of these insights the following approach was applied (the implementation has not been fully sequential since some of the steps were performed in parallel and also iterations were necessary):

1. Acquisition of relevant databases. Factors considered when selecting the set of databases were: availability, price, coverage (sectors, time, geographical area) and quality. Among databases which apparently were very similar and more or less totally overlapping, only the most representative one was selected. Databases containing accidents for one specific country only, were of lower interest. Chapter 2.2 discusses the major databases consulted as information sources when establishing PSI's database ENSAD.

2. Implementation of the acquired databases on a PC. User requirements concerning the overall database were relatively moderate since the final product was intended exclusively for internal uses at PSI and not for external distribution.

3. Merging of the contents of the various databases within the framework of Microsoft's Access Database. In view of the focus on energy-related accidents, not all information was retained when merging the databases into a single structure.

4. Elimination of overlapping events and/or harmonisation of non-consistent information. The latter required consultation of sources beyond the available databases (see also point 7 below).
5. Identification of energy-related accidents and among them of accidents considered as severe (see chapter 3.1 for the defintion of severe accident as used in this study).
6. Allocation of energy-related accidents to specific fuel cycles and subsequently to specific stages within each fuel cycle.
7. Searches utilising supplementary sources of information and aiming at checks and identification of additional events; analysis of the assembled material. This includes: annual publications, general and specialised literature, national and international newspapers, incident lists and reports, and direct contacts with responsible companies and other competent organisations or individuals. Such investigations are extremely time and resource consuming. For this reason within the present effort checks and complementary analyses beyond the main sources of information were concentrated on events which have very severe consequences and/or are subject to major uncertainties with respect to the real extent of consequences. Particular attention has been given in this context to the applicability and transferability of the data.
8. Implementation of the additional evidence into the database. Given that new events have been identified this includes also the steps under points 5 and 6 above.
9. Evaluations based on the “final” set of data. The evaluations of severe accident frequencies and various types of consequences were first carried out for each energy carrier. These results were then used for comparisons between the various energy sources. The results were normalised on the basis of energy production by means of each of the sources.

2.2 Information sources

In the past, significant efforts have been directed towards the development of databases for historical events with the purpose of understanding the potential hazards confronting industrial designers, insurance companies or decision makers. Efficient risk management and hazard control can be defined and implemented if lessons are learned from previous incidents and accidents (ICOLD, 1974; Baecher et al., 1980; Drogaris, 1983; Beek, 1994). The experience gained from the analysis of past accidents can be used to avoid design errors, to improve existing facilities, to develop emergency plans, to evaluate specific technologies, etc.

PSI’s database ENSAD was used as a basis because at the start of the NewExt Project it has already integrated historical data from a large variety of sources up to year 1996 (Hirschberg et al., 1998). Thus, the statistical evidence available for fossil systems is very extensive and can be regarded as quite satisfactory for comparative studies. Nevertheless, specific tasks were pursued aiming at extensions of the database and at creating a basis for evaluations consistent with the objectives of this work package. Specifically:

- Numerous relevant external database inputs with respect to suppliers, scope, update frequency, costs etc. were reviewed.
- Table 1 shows the major commercial and non-commercial databases consulted as information sources that were used to update and extend ENSAD within this project. In addition to databases covering a broad spectrum also specific databases were searched. For example, ICOLD (International Committee on Large Dams) and “Bibliography of the History of Dam Failures” were used for dam accidents, and WOAD and ITOPF for oil spills.
- For normalization the continuously updated IEA database on world-wide energy production and consumption was used (IEA, 2002).
- Searches for historical data on small accidents were initiated. Though these accidents were not in focus, they were addressed on a much lower level of detail, in order to put severe accidents into perspective. First analyses indicated that small accidents are strongly underrepresented in the available databases.
- Development of a concept for experience-based dam risk assessment, if feasible, with stronger consideration of design- and location-specific factors.

Table 1: Major accident databases of relevance used for PSI's ENSAD. Databases marked in bold have been used as major information sources for the NewExt extension of ENSAD; whereas other databases were of minor importance because they contributed little additional information and/or were discontinued. The time period refers to the currently available databases; the actual period considered when using them as information sources within ENSAD may be different in some cases.

Full name of the database (contact organisation or originators)	Country of origin	Database code name	Time period	Geogra-phical area	Accidents covered
OFDA/CRED International Disaster Database [a]	USA	**EM-DAT**	1900-2003	World-wide	Man-made and natural catastrophes
Hazards Incidence Data Service of the UK Health and Safety Executive (HSE) [b]	UK	**MHIDAS**	1900-2003	World-wide	Industrial accidents
Library and Information Services of the UK HSE [b]	UK	**HSELINE** [c]	1900-2003	World-wide	Accidents related to health and safety at work
The Failure and Accidents Technical Information System (TNO)	Netherlands	**PC-FACTS**	1900-2003	World-wide	Industrial accidents
The "SIGMA" Publication Swiss Re Company	Switzerland	**SIGMA**	1969-2003	World-wide	Man-made and natural catastrophes
Lloyd's Casualty Week (LLP, formerly Lloyd's of London Press)	UK	**LLP** [c]	1976-2003	World-wide	Industrial accidents
The World-wide Offshore Accident Databank (DNV)	Norway	**WOAD**	1970-1998	World-wide	Offshore accidents
International Tanker Owners Pollution Federation Limited	UK	**ITOPF**	1974-2003	World-Wide	Tanker accidents
ETC Tanker Spills Database (US Department of the Interior, Minerals Management Service)	USA	**ETC**	1974-1997	World-Wide	Tanker accidents
The ICOLD Catalogues of Dam Disasters (ICOLD)	France	**ICOLD**	1850-2000	World-wide	Dam accidents
Bibliography of the History of Dam Failures	Austria	**BHDF**	2500 b.C.-2001	World-wide	Dam accidents
China Coal Industry Yearbook	China	**CCIY**	1994-2000	China	Coal chain accidents
The Fatal Hazardous Materials Accidents Database (RfF)	USA	RfF	1945-1991	World-wide	Man-made and natural catastrophes
The Accident Handbook (UBA)	Germany	Handbuch Störfälle	1900-1986	World-wide	Industrial accidents
The Major Accident Reporting System (CEC JRC-Ispra)	European Community	MARS	1980-1991	Europe	Industrial accidents
Book of the Year (Encyclopaedia Britannica)	UK	Encyclopaedia Britannica	1973-1997	World-wide	Man-made and natural catastrophes
Catalogue of Dam Disasters, Failures and Accidents (Babb and Mermel)	USA	CDDFA	1800-1968	World-wide	Dam accidents
Study on Large Losses in the Gas and Electric Utility Industry (Marsh & McLennan)	USA	MM	1965-1990	World-wide	Accidents in gas and electric utility industry
Minerals Management Service Database (access through WOAD)	USA	MMS	1970-1989	USA	Offshore accidents
Acute Hazardous Event Database (EPA)	USA	AHE	1900-1985	USA	Chemical accidents
SONATA Database (TEMA/ENI)	Italy	SONATA	1850-1998	World-wide	Industrial accidents
VARO Databank (FIOH)	Finland	VARO	1978-1998	Finland	Man-made and natural catastrophes

[a] CRED: WHO Collaborating Centre for Research on the Epidemiology of Disasters; OFDA: The US Office of Foreign Disaster Assistance Database
[b] MHIDAS and HSELINE are part of OSH-ROM (SilverPlatterDirectory, 2003). OSH-ROM also contains the databases NIOSHTIC and NIOSTHIC-2 (Bibliographic database published by the US National Institute of Occupational Safety and Health) and CISDOC (a product of the International Occupational Health and Safety Centre (CIS) of the International Labour Organisation (ILO).
[c] Note that in this report the databases HSELINE and LLP are not treated as separate sources because most accidents found in LLP were also included in HSELINE. Thus, HSELINE/LLP was used as a common database code name.

3. Structure and content of ENSAD

3.1 Severe accident definition

Based on the literature, there is no unique definition of a severe accident. All definitions include various consequence (damage) types (evacuees, injured persons, fatalities or costs) and a minimum level for each damage type. The differences between the definitions concern both the set of specific consequence types considered and the damage threshold.

This can be illustrated by the following examples. The ‘‘World-wide Offshore Accident Database’’ (WOAD) of the Det Norske Veritas (DNV, 1999) considers an accident as severe or major, if more than one fatality occurred or if the damaged unit (e.g., oil platform, drill ship or drill barge) experienced total loss. Glickman & Terry (1994) define a significant accident for technological hazard, if it resulted in at least 5 fatalities or if it involved the release of a chemical, petroleum product, hazardous waste or other hazardous material. The SIGMA publication series of Swiss Re Company (Swiss Re, 2001) and Rowe (1977) do not use the term “severe accidents”. However, they do investigate and collect data on catastrophic events. The criteria are arbitrary (Rowe, 1977), not standardized and can be adjusted with time (Swiss Re, 2001). As another example, EM-DAT (EM-DAT, 2003) considers several reasons for taking into account a disaster, e.g., 10 or more people killed, 100 or more people affected/injured/homeless, declaration of state of emergency, disaster that affected several countries/regions.

The PSI database ENSAD uses seven criteria to define a severe accident:

1) at least five fatalities or
2) at least ten injured or
3) at least 200 evacuees or
4) extensive ban on consumption of food or
5) releases of hydrocarbons exceeding 10’000 t or
6) enforced clean-up of land and water over an area of at least 25 km^2 or
7) economic loss of at least five million USD(2000).

Whenever any one of the above criteria is satisfied, the accident is considered to be severe. However, various types of consequences are covered to differing extents because of differences in availability and quality of information. The highest degree of completeness is available for fatalities, whereas information for injured and evacuees is often more uncertain. For example, the database MHIDAS states the number of evacuated people as “> x” if only an approximate number is available, and in the database EM-DAT “thousands of homeless” may translate to “2000 homeless” (although it is probably underestimated). Thus fatalities provide the most complete and reliable damage indicator listed above.

The damage indicator “economic costs” is to some extent inconsistent because different sources report different types of economic loss. Insured losses (or damages) provide a particularly suitable basis for analyses as they can be established precisely, but their availability is largely restricted to accident records compiled by reinsurance companies, such as Swiss Re or Munich Re. Economic losses on the other hand can never be calculated exactly as they are determined in various ways, depending on the definition applied in each case, and are seldom fully and reliably established (e.g., Munich Re, 2001). Furthermore, they can consist of direct losses (immediately visible, countable losses), indirect losses (resulting from the physical destruction of assets) and secondary costs (costs that weaken the affected country’s economy); however, the components considered are often not clearly stated. Economic losses are also sometimes called “total loss” (or damage) - including insured and uninsured damage - (Swiss Re, 2001) or “estimated damage” (EM-DAT, 2003).

Releases of hydrocarbons are almost exclusively used in connection with oil spills, but is of little relevance for the other energy chains considered. The indicators dealing with ban on consumption of food and enforced clean-up of land and water were not further addressed in accident analysis as the ENSAD database contains only very few entries with information on these damage categories (also compare, Hirschberg et al., 1998).

While the focus of this work is on severe accidents, smaller (or minor) accidents are also addressed in order to provide a broad perspective on the accident issue. A smaller accident does not meet any of the criteria established for definition of a severe accident. Given that the coverage of smaller accidents is much more incomplete due to underreporting, they are only treated at a relatively coarse level of detail.

3.2 Some facts about ENSAD

3.2.1 Overall statistical information of ENSAD

Figure 1 provides an overview of the number of accidents by type and by different damage categories, as included in ENSAD. Man-made accidents comprise 12’943 or 70.3%, whereas natural disasters amount to 5457. A total of 6404 energy-related accidents corresponds to 34.8% of all accidents or 49.5% of man-made accidents. Among the energy-related accidents 3117 (48.7%) are severe, of which 2078 have 5 or more fatalities. One should note that non-energy-related accidents and natural disasters are a second priority within ENSAD. Consequently, the corresponding data are likely to be less comprehensive than the ones provided for the energy-related accidents.

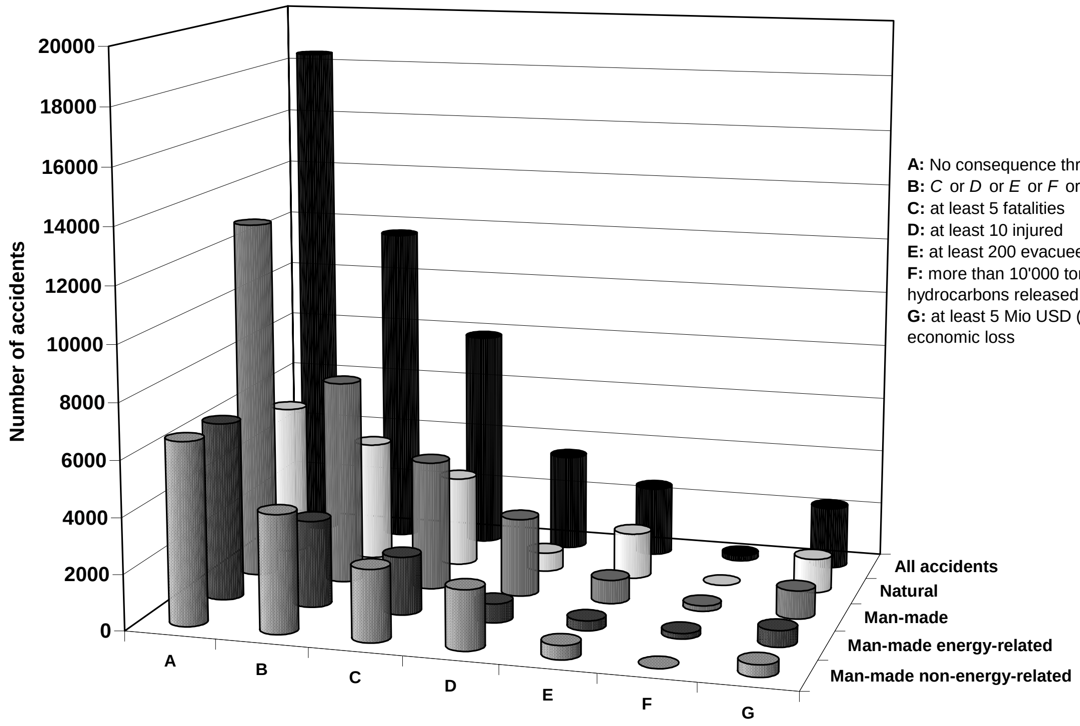

Figure 1: Overview of the number of accidents by type (natural, man-made, man-made energy-related, man-made non-energy-related) and by different damage categories (indices A-G), as included in ENSAD.

Of the 18'400 accidents contained in the ENSAD database, about 89% occurred in the period 1969-2000 (Figure 2). Data for 2001-2003 are not fully representative, as there is normally a certain time lag in reporting and the publication of the data. Therefore, forthcoming results and statistical evaluations presented in this report focus on the period 1969-2000. Additonally, the number of energy-related accidents, on which this task focuses, exhibited a distinct increase since the late sixties (also compare Hirschberg et al., 1998), giving additional support to this selection.

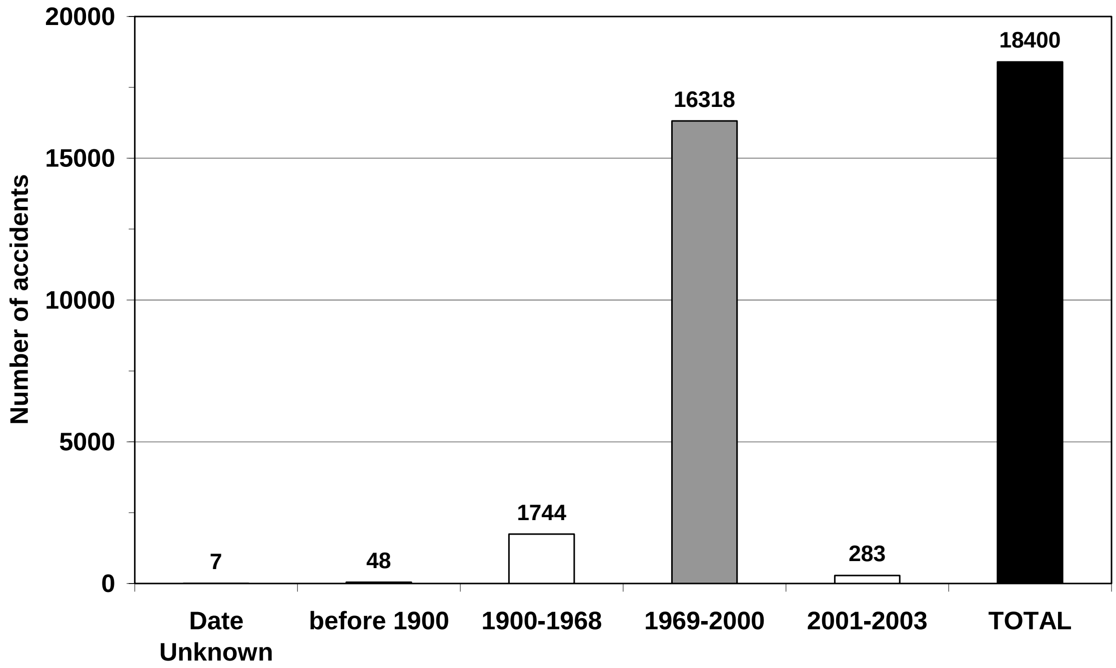

Figure 2: Number of accidents in ENSAD for various time periods.

3.2.2 Distribution of severe accidents by various categories

The distribution of severe (≥ 5 fatalities) accidents for the years 1969-2000 is shown in Figure 3 (number of accidents) and Figure 4 (number of fatalities). Since the early 1990's, the number of severe accidents has significantly increased for natural disasters and man-made accidents[1]; values for 1993 to 2000 being distinctly higher than the average.

The much higher numbers of man-made energy-related, severe accidents for the years 1994 to 1999 (with peaks in 1995 and 1997) is due to the availability of detailed statistics for the Chinese coal chain, as reported in the China Coal Industry Yearbook (CCIY, 1997-2000). The distinct decrease in year 2000 is due to lack of the corresponding data from this information source at the present time. Incompleteness of the ENSAD data for China was earlier recognized (Hirschberg et al., 1998). Integration of data from the Coal Industry Yearbook allows a reasonably accurate assessment. However, this does not apply to years before 1994 because data for China were of rather poor quality in terms of completeness and consistency.

Number of fatalities were significantly higher for natural disasters than man-made accidents. However, values exhibit large annual fluctations because great catastrophes have a strong influence. The largest natural disasters were a storm and flood catastrophe in

1 In recent years, information has become more easily available, especially in developing countries, which in part compensates for the observed increase in the number of severe accidents; particularly for man-made accidents.

Bangladesh in 1970 (300’000 fatalities), the Tangshan earthquake in China in 1976 (290’000), and a drought and civile strife in Sudan in 1983 (250’000). The three largest man-made energy-related accidents resulted in fatalities one order of magnitude lower. In 1975, the Banqiao/Shimantan dam failure in China caused 26’000 fatalities; in 1987, the collision of a passenger ferry and an oil tanker off the Philippines resulted in 4375 fatalities; and in 1982, 2700 soldiers and civilians died in the collision of a soviet fuel truck and another vehicle in the Salang tunnel (Afghanistan). Fore more information on the most severe energy-related historical accidents compare Tables 4 to 7 in chapter 5.2.

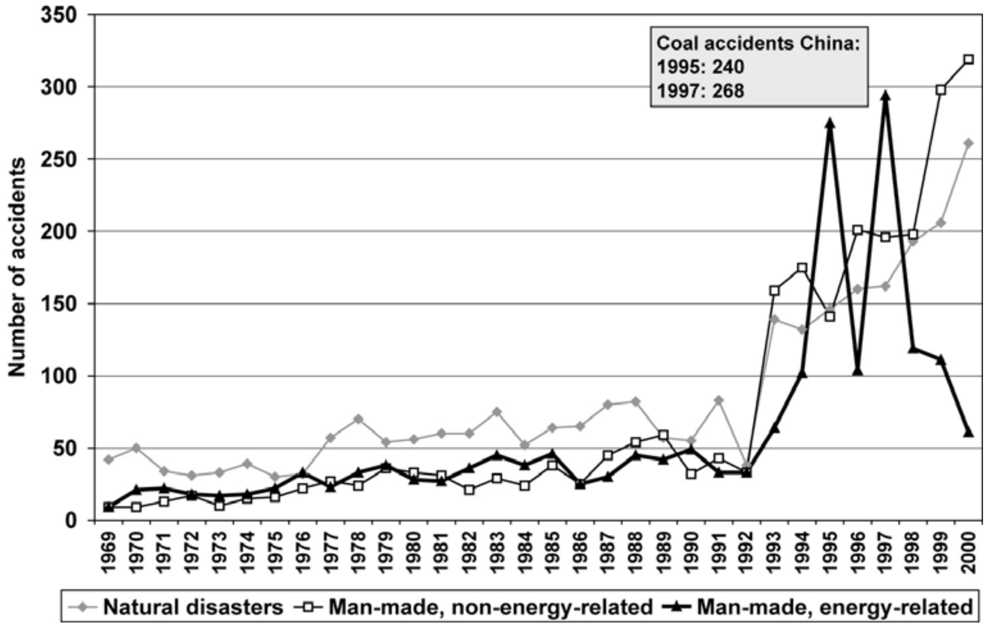

Figure 3: Number of severe (≥ 5 fatalities) accidents that occurred in natural disasters and man-made accidents in the period 1969 to 2000.

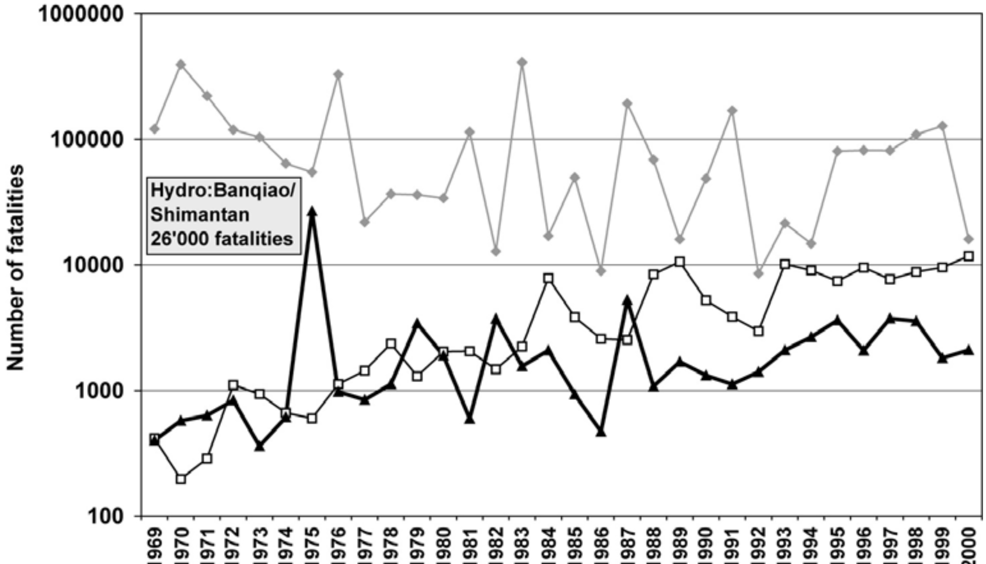

Figure 4: Number of fatalities in severe (≥ 5 fatalities) accidents that occurred in natural disasters and man-made accidents in the period 1969 to 2000.

Finally, severe (≥ 5 fatalities) accidents per contient are given in Figure 5. Both, natural disasters and man-made, non-energy related accidents were dominated by Aisa. For man-made, energy-related accidents, a similar trend is observed, but only if accidents in the Chinese coal chain are fully accounted for. Otherwise, the share of Asia would drop to about 37%, and shares of America and Europe increase up to 27% and 25%, respectively.

Asia has been especially hard hit by natural disasters because the region is large and heavily populated, particularly in dangerous coastal areas. Furthermore, there is frequent seismic, tropical storm, and flood activity. Asia's natural and social vulnerability concerning natural disasters also has been recognized in other studies (e.g., Abramovitz, 2001). In contrast, distribution patterns for man-made accidents are more affected by differences in level of industrialization and safety standards.

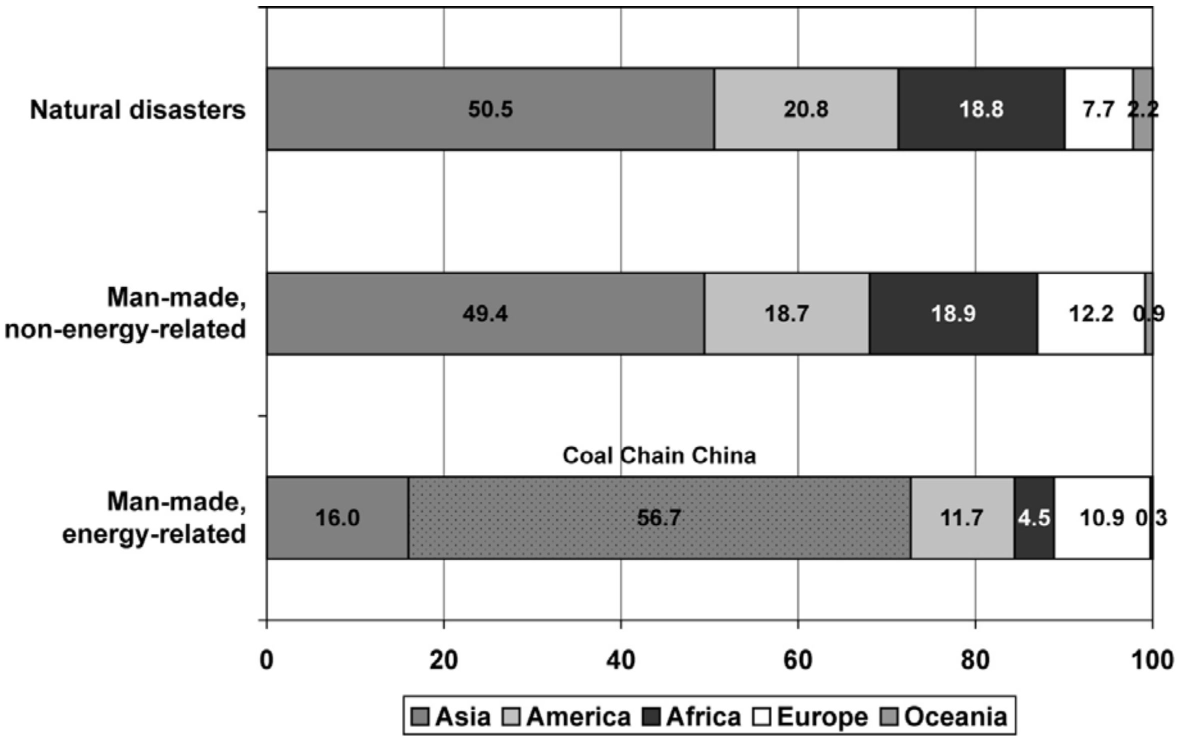

Figure 5: Distribution by continent of severe (≥ 5 fatalities) accidents that occurred in natural disasters and man-made accidents.

4. Principles and assumptions for evaluation

4.1 Energy chain stages

The risks to the public and the environment, associated with various energy systems, arise not only at the power plant stage but at all stages of energy chains. In general, an energy chain may comprise the following stages: exploration, extraction, transport, storage, power and/or heat generation, transmission, local distribution, waste treatment, and disposal. However, one should be aware that not all these stages are applicable to every energy chain. Table 2 gives an overview of distinct stages for the coal, oil, gas, nuclear and hydro chains.

Table 2: Stages of different energy chains.

	Coal	Oil	Natural Gas	LPG	Nuclear	Hydro
Exploration	Exploration	Exploration	Exploration	--	Exploration	
Extraction	Mining and Coal Preparation	Extraction	Extraction and Processing	--	Mining / Milling	
Transport	Transport to Conversion Plant	Transport to Refinery (Long Distance Transport)	Long Distance Transport (pipeline)	--	Transport	
Processing	Conversion Plant	Refinery		• Refinery • Natural gas processing Plant	Upstream Processing [a]	
Transport	Transport	Regional Distribution	Distribution: • Long Dist. • Regional • Local	Distribution : • Long Dist. • Regional • Local	Transport [b]	
Power / Heat Generation	Power Plant Heating Plant	Power Plant Heating Plant	Power Plant Heating Plant	Heating Plant	Power Plant	Power Plant
Transport					Transport to Reprocessing Plant	
Processing					Reprocessing	
Waste Treatment	Waste Treatment				Waste Treatment	
Waste Disposal	Waste Disposal				Waste Disposal	

[a] Includes: Conversion, Enrichment, Fuel Fabrication.
[b] Includes transports between the processing stages mentioned in note a.

4.2 Allocation of damages

In chapter 6 aggregated, normalized, energey-related severe accident records are compared. In this context distinction is made between OECD and non-OECD countries[2]. OECD countries import from non-OECD countries a large fraction of their total consumption of crude oil and LPG, a small fraction of natural gas and a negligible fraction of coal. The net import from non-OECD countries is negligible for hydro and nuclear power.

A difficulty that arises in comparative studies with aggregated normalized severe accident records is that a large number of severe accidents occurs in non-OECD countries at stages in the energy chain relevant for the export to OECD countries. The relevant stages are “Exploration”, “Extraction” and “Transport to the Refinery” for oil; “Exploration”, “Extraction” and “Long Distance Transport” for natural gas; and “Refinery”, “Natural Gas Processing Plant” and “Long Distance Transport for LPG. In comparative studies apart from the straight-forward case with no reallocation of accidents, of interest is also the alternative scheme with appropriate share of the consequences of accidents that occurred at such fuel cycle stages in non-OECD countries being added to the damages which physically occurred in OECD countries; the net amounts of energy carriers imported to OECD countries from non-OECD countries form the basis for this allocation. A detailed description of the allocation procedure used can be found in Hirschberg et al. (1998).

It should be noted that no allocation scheme has been used in the estimations of external costs; these are given for OECD and non-OECD separately. Furthermore, it is in principle feasible to estimate the external costs for a given fuel chain configuration based on the combination of region-specific external costs for the individual fuel chain stages. Such estimates are, however, subject to large uncertainties due to the statistical scarcity of the decomposed data material.

4.3 Econometric valuation of severe accidents

The principal objective of the econometric analysis is to derive unit values that express the welfare impacts of accidents in the non-nuclear energy supply chain in monetary terms, and enable calculation of the external costs of such accidents. Thus, for a given welfare impact unit, (e.g., a work-place injury), we look to identify a monetary value that represents the willingness to pay (WTP) to avoid the impact or the willingness to accept (WTA) compensation to bear the injury. A taxonomy of external cost impacts that might result from a major accident in energy chains includes:

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[2] Note that aggregated results are also presented for EU15, but allocation procedures used for OECD- and non-OECD countries were not feasible.

- Mortality (with or without hospitalisation) in accident
- Morbidity - physical injury in accident
- Mental trauma - from physical injury, evacuation
- Evacuation (costs of resettlement/accommodation)
- Clean-up/repair costs and willingness to pay (WTP) for recreational/ ecosystem losses - oil spills
- Ban on consumption of food
- Land contamination
- Other economic losses

For a detailed treatment of monetization of severe accidents compare chapter 7.

5. Severe energy-related accidents worldwide

5.1 Severe accident database

Available information on severe accidents in non-nuclear energy chains is summarized in Table 3. Evaluations and analyses were focused on (but not limited to) fatalities because information on other indicators such as injured, evacuees or economic costs were not available at a comparable level of completeness. However, aggregated indicators could still reveal some general trends.

Table 3: Summary of the severe accident database for accidents with at least five fatalities. The time period considered is 1969 – 2000. Accident statistics are given for the categories OECD (incl. EU15), EU15 alone, and non-OECD.

	OECD		**EU15**		**non-OECD**	
Energy chain	*Accidents*	*Fatalities*	*Accidents*	*Fatalities*	*Accidents*	*Fatalities*
Coal	78	2259	11	234	102 1044 [(a)]	4831 18'017 [(a)]
Oil	165	3789	58	1141	232	16'494
Natural Gas	80	978	24	229	45	1000
LPG	59	1905	19	515	46	2016
Hydro	1	14	0	0	10	29'924 [(b)]

[(a)] First line: Coal non-OECD w/o China; second line: Coal China

[(b)] Banqiao and Shimantan dam failures together caused 26'000 fatalities

5.2 Most severe historical accidents

Tables 4 to 7 provide lists of the ten worst accidents in the period 1969-2000 within the damage categories "immediate fatalities", "injured", "evacuees" and "costs". While one specific indicator (shown in bold face) is in focus of each table, also other parameters characterizing the consequences are given. Note that "latent fatal and non-fatal cancers", particularly relevant for the Chernobyl accident, constitute a separate category, which is not included in the tables. For all indicators, whenever a range of values is available for a specific damage category only the highest number is provided in the table.

Considering fatalities, nine out of the ten most severe accidents occurred in non-OECD countries; and were attributable to oil and hydro chains (Table 4). For injured shares for OECD and non-OECD countries were about equal, with the majority of accidents taking place in the oil and LPG chains (Table 5). The ratio of OECD and non-OECD countries for evacuees was similar to injured, but in contrast to fatalities and injured no oil chain accidents were in the top three accidents with highest number of evacuees (Table 6). Highest costs occurred primarily in OECD countries, with exception of Chernobyl (Table 7). The speculative and highly uncertain costs for the Chernobyl accident were estimated two orders of magnitude higher than any other accident. The Exxon Valdez oil spill (Prince William Sund, Alaska) was the most costly non-nuclear accident.

The provided cost estimates have been rounded. For the discussion of (lacking) consistency we refer to the discussion in (Burgherr et al., to be published).

Table 4: Ten energy-related severe accidents with the highest number of immediate fatalities in the period 1969-2000.

Date	Country	Energy chain	Energy chain stage	Fatalities	Injured	Evacuees	Costs (Mio USD 2000)
05.08.1975	China	Hydro	Power Plant	26'000	—	—	—
20.12.1987	Philippines	Oil	Transport to Refinery	4375	26	—	—
01.11.1982	Afghanistan	Oil	Regional Distribution	2700	400	—	—
11.08.1979	India	Hydro	Power Plant	2500	—	150'000	1260
18.09.1980	India	Hydro	Power Plant	1000	—	—	—
18.10.1998	Nigeria	Oil	Regional Distribution	900	100	—	—
04.06.1989	Russian Federation	LPG	Long Distance Transport	600	755	—	—
02.11.1994	Egypt	Oil	Regional Distribution	580	>1	20'000	160
29.06.1995	Republic of Korea	Oil	Regional Distribution	577	952	—	—
25.02.1984	Brazil	Oil	Regional Distribution	508	150	2500	—

Table 5: Ten energy-related severe accidents with the highest number of injured in the period 1969-2000.

Date	Country	Energy chain	Energy chain stage	Fatalities	Injured	Evacuees	Costs (Mio USD 2000))
19.11.1984	Mexico	LPG	Regional Distribution	498	7231	250'000	3
17.01.1980	Nigeria	Oil	Extraction	180	3000	—	—
22.04.1992	Mexico	Oil	Regional Distribution	252	1600	5000	370
04.10.1988	Russian Federation	Oil	Regional Distribution	5	1020	—	—
19.12.1982	Venezuela	Oil	Power Plant	160	1000	40'000	93
25.01.1969	USA	LPG	Regional Distribution	2	976	100	14
29.06.1995	Republic of Korea	Oil	Regional Distribution	577	952	—	—
05.06.1976	USA	Hydro	Powe Plant	14	800	35'000	2720
01.07.1972	Mexico	LPG	Regional Distribution	8	800	300	5
04.06.1989	Russian Federation	LPG	Long Distance Transport	600	755	—	—

Table 6: Ten energy-related severe accidents with the highest number of evacuees in the period 1969-2000.

Date	Country	Energy chain	Energy chain stage	Fatalities	Injured	Evacuees	Costs (Mio USD 2000)
19.11.1984	Mexico	LPG	Regional Distribution	498	7231	250’000	3
11.11.1979	Canada	LPG	Regional Distribution	—	8	250’000	24
28.03.1979	USA	Nuclear	Power Plant	—	—	200’000	5960
11.08.1979	India	Hydro	Power Plant	2500	—	150’000	1260
14.09.1997	India	LPG (OIL)	Refinery	60	39	150’000	27
26.04.1986	Ukraine	Nuclear	Power Plant	31	370	135’000	372’300
25.05.1988	Mexico	Oil	Regional Distribution	—	7	100’000	—
26.02.1988	USA	Oil	Regional Distribution	>1	—	60’000	2
19.12.1982	Venezuela	Oil	Power Plant	160	1000	40’000	93
05.06.1976	USA	Hydro	Power Plant	14	800	35’000	2720

Table 7: Ten energy-related severe accidents with the highest monetary damages in the period 1969-2000. Costs are expressed in million USD(2000). Note that the cited costs are in many cases very uncertain and due to differences in the definitions subject to major inconsistencies. Compare also chapter 3.1 and the discussion provided in Burgherr et al. (to be published).

Date	Country	Energy chain	Energy chain stage	Fatalities	Injured	Evacuees	Costs in Mio USD(2000)
26.04.1986	Ukraine	Nuclear	Power Plant	31	370	135’000	372’300
28.03.1979	USA	Nuclear	Power Plant	—	—	200’000	5960
24.03.1989	USA	Oil	Transport to Refinery	—	—	—	2780
05.06.1976	USA	Hydro	Power Plant	14	800	35’000	2720
28.01.1969	USA	Oil	Extraction	—	—	—	2630
07.07.1988	UK	Oil	Extraction	167	—	—	2180
02.01.1997	Japan	Oil	Transport to Refinery	1	—	—	1320
25.09.1998	Australia	Natural Gas	Extraction (Processing)	2	8	120	1296
11.08.1979	India	Hydro	Power Plant	2500	—	150’000	1260
26.07.1996	Mexico	Natural Gas	Extraction (Processing)	9	47	—	1100

5.3 Distribution of severe energy-related accidents by years

On average, 58 energy-related accidents with at least five fatalities occurred each year world-wide (Figure 6). About 60% of all accidents happened in the period 1993-2000. This dominance is primarily due to improved reporting of coal chain accidents in China and their publication in the China Coal Industry Yearbook (CCIY). Considering different gravity indices for fatalities, over 72% of all accidents resulted in 5-20 fatalities; whereas accidents exceeding 100 fatalities ranged between 0 to 5 per year.

The average number of fatalites was 2539 per year, but would drop to about 1727, if the largest accident (Banqiao/Shimantan dam failure with 26’000 fatalities) is excluded (Figure 7). The influence of the availability of data from the CCIY is also evident for fatalities. In contrast to number of accidens, several peaks can be observed for fatalities in years before 1993, which are attributable to single large events (compare Table 4).

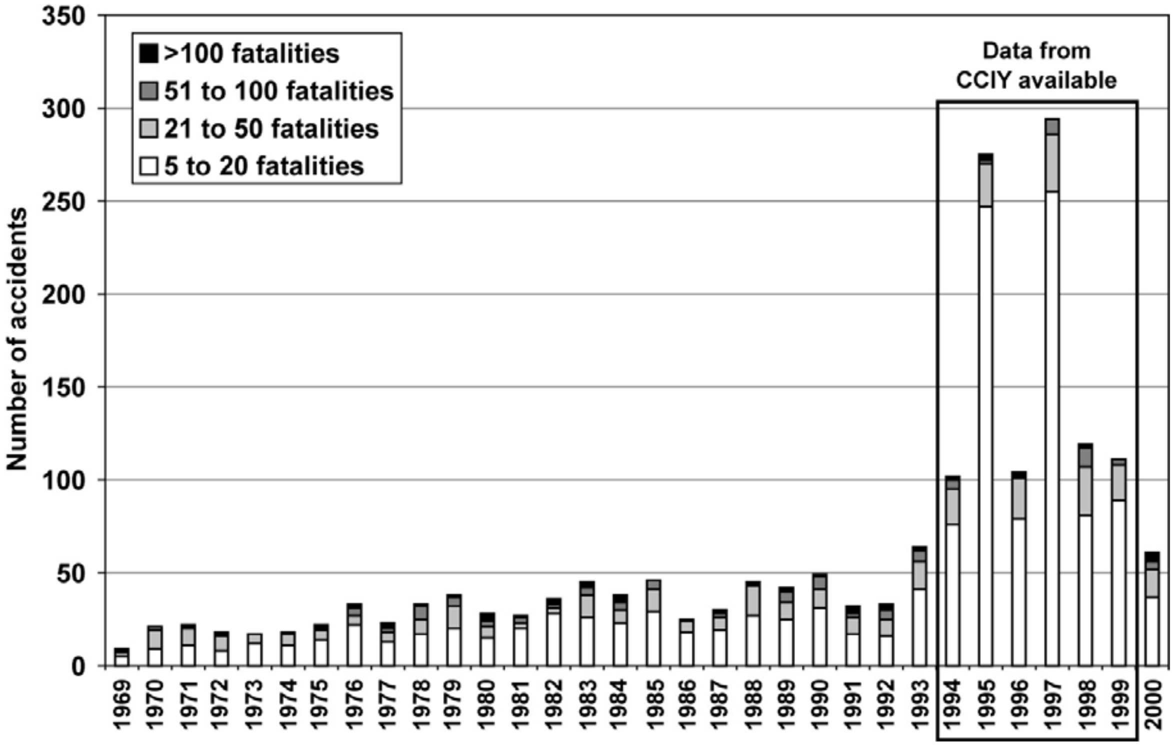

Figure 6: Severe energy-related accidents world-wide during the peiod 1969-2000, with different gravity indices for fatalities. The rectangular box indicates the time period for which extensive data of the China Coal Industry Yearbook (CCIY) were available.

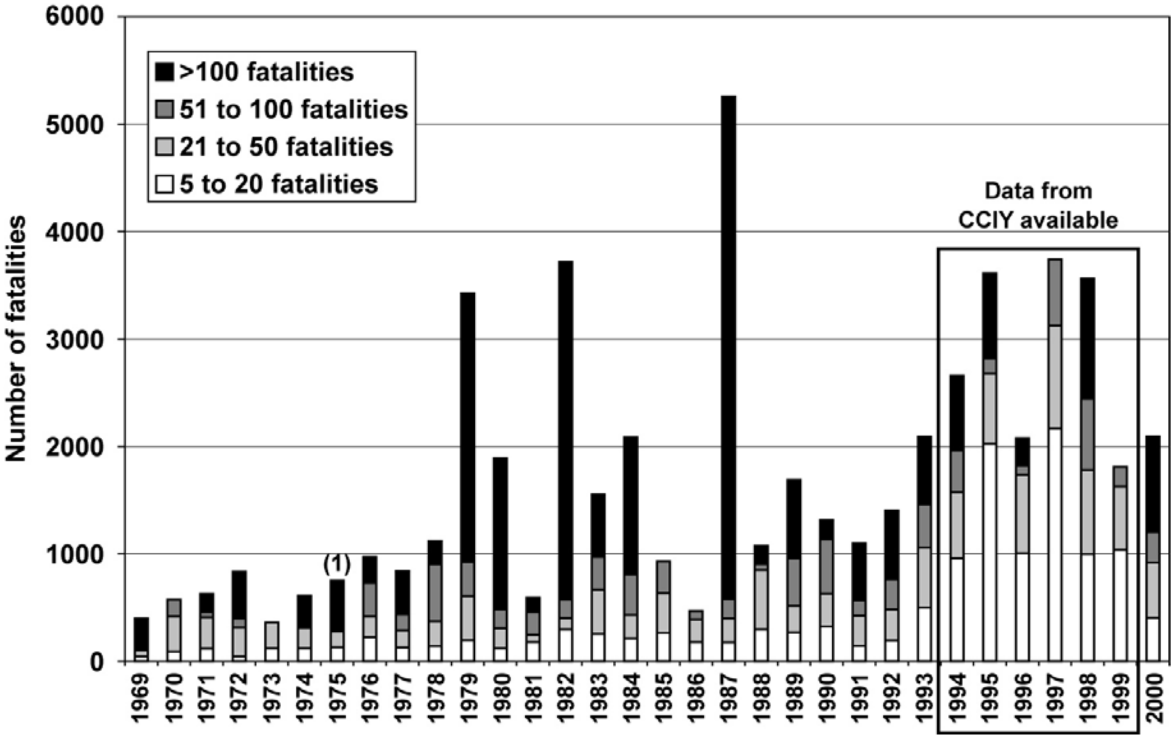

Figure 7: Severe energy-related accident fatalities world-wide during the period 1969-2000, with different gravity indices for fatalities. The rectangular box indicates the time period for which extensive data of the China Coal Industry Yearbook (CCIY) were available.
(1): Data for Banqiao/Shimantan dam failure (26’000 fatalities) not shown for graphical reasons.

5.4 Severe vs smaller accidents

The term “smaller accident” is used for those accidents that do not fulfil any of the criteria used to define a severe accident, as described in chapter 3.1. These accidents were not in focus of the study and could not be addressed on the same level of detail. However, searches for historical data on smaller accidents were also carried out to put severe accidents into perspective. The survey performed indicates that the completeness of reporting is correlated to the severity of accidents, i.e. the lower the damage the higher the likelihood that the accident will not be found in the databases considered. The findings were also indicative that indicators other than fatalities were even much more incomplete than in the case severe accidents. Therefore, the results presented here are primarily based on fatalities.

Two categories of smaller accidents can be distinguished. “True” smaller accidents have less than 5 fatalities and are also not exceeding any other criteria used to define a severe accident. “Partial” smaller accidents have less than 5 fatalities but according to the other criteria they receive the status of a severe accident, and are also incorporated in the respective evaluations. This group accounts for 25% of the number of accidents (Figure 8) and about one third of the number of fatalities (Figure 9) shown here. Finally, a third group of smaller accidents (0 fatalities, but at least one other indicator >0 and below the threshold for a severe accident) was not included because indicators often lacked precision (e.g., >1 evacuees without giving an upper range).

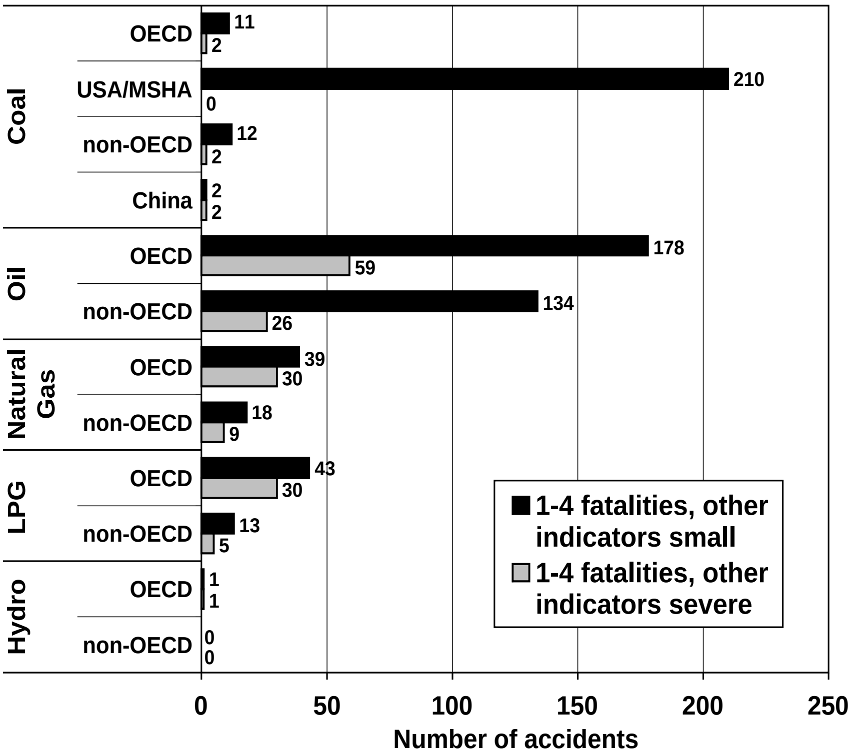

Figure 8: Number of smaller accidents in OECD and non-OECD countries for the period 1969-2000. Categories represent "true" smaller accidents and "partial" smaller accidents (see text for explanations).

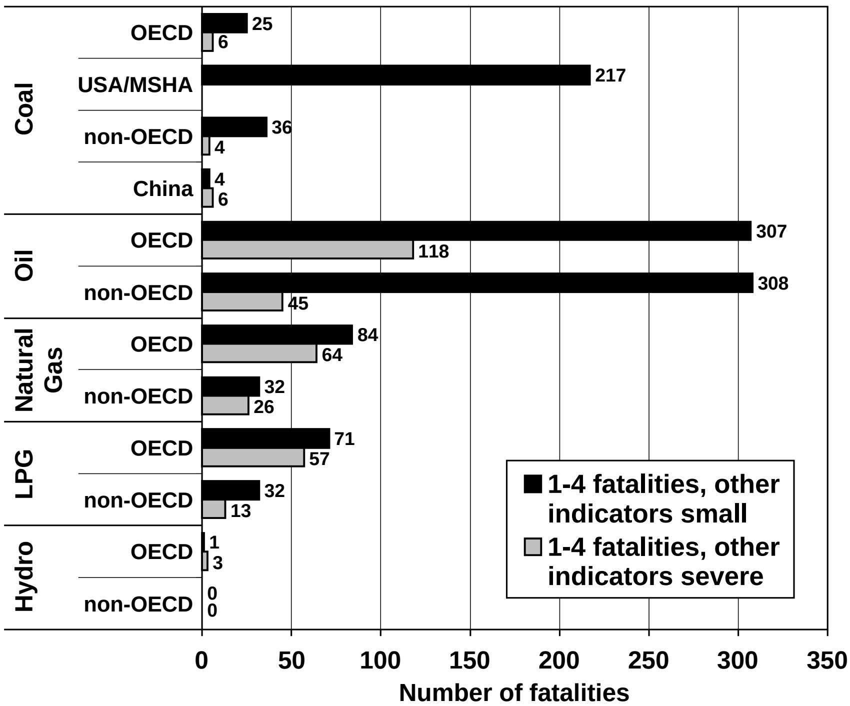

Figure 9: Number of fatalities in smaller accidents in OECD and non-OECD countries for the period 1969-2000. Categories represent "true" smaller accidents and "partial" smaller accidents (see text for explanations).

Nevertheless, there are few exceptions of the general poor representation of smaller accidents; i.e., comprehensive statistics exist for the US and Chinese coal chains. In the period 1995-2000, there was a total of 210 smaller accidents resulting in 217 fatalties in the US coal chain, but not a single severe accident. In contrast, accidents in China's coal chain resulted on average in 6200 fatalities per year for the period 1994-1999, of which 30% were severe accidents and 70% smaller accidents.

In conclusion, data of smaller accidents appear to be clearly underestimated due to significant underreporting, although some exceptions were recorded. Additional bias is attributable to sometimes substantial shortcomings in data accuracy for indicators other than fatalities.

6. Energy chain comparisons

6.1 Occupational vs public accidents

Severe accidents in the energy sector are often work-related, but can also affect the general public. For example, consequences of coal mine accidents are mostly restricted to the workers that are present at the time of the accident, although rescue parties may be at risk as well. In contrast, failures of hydro dams could have large effects on downstream residents. In many cases, however, an accident may not be exclusively allocated to one or the other category.

Separation of public and occupational accidents is also an important prerequisite for subsequent econometric analyses (see chapter 7) because degrees of internalization substantially influence the transfer from damage costs to external costs.

Figure 10 shows the percent shares of occupational and public fatalities for the different energy chains in OECD and non-OECD countries, as established in the present project. With very few exceptions, fatalities in the coal chain accidents are work-related. For the oil chain, OECD countries exhibit about equal shares for occupational and public fatalities, whereas the latter accounted for more than 80% in non-OECD countries., This is largely due to the two very large accidents in Afghanistan (1982) with 2700 fatalities and the Philippines (1987) with 4375 fatalities (see section 6.2 for details). In natural gas and LPG accidents, public fatalities amounted roughly to 60% and 80%, respectively. Floods resulting from failures of large hydro dams are primarily affecting downstream settlements, i.e., the general public.

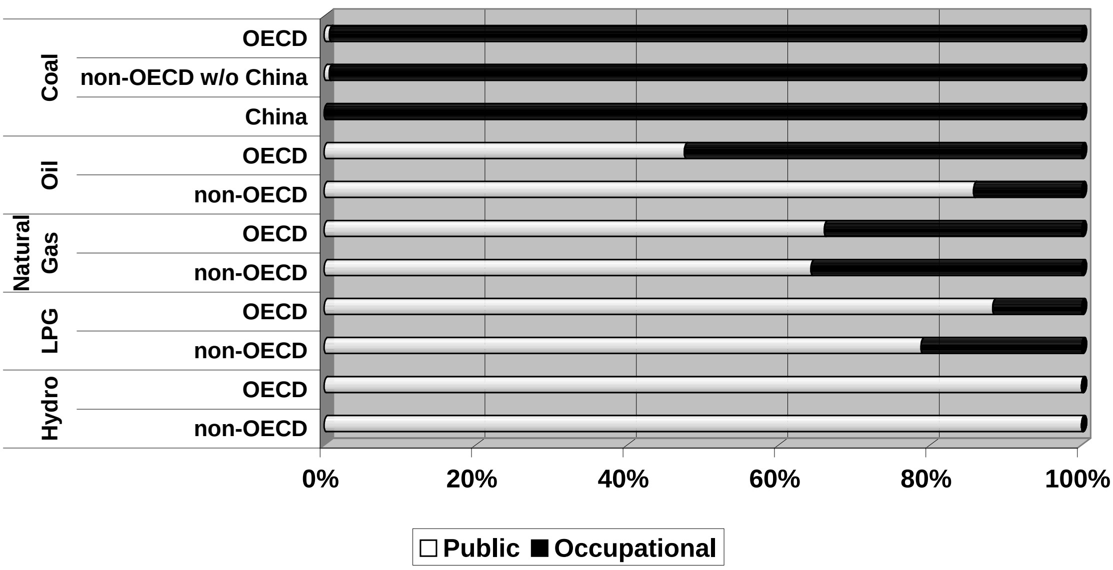

Figure 10: Shares of occupational and public fatalities attributable to the different energy chains for the period 1969-2000.

6.2 Aggregated indicators and frequency consequence curves

The evaluations presented in the following address different severe accident indicators such as the number of accidents, fatalities, injured, evacuees and the extent of monetary damages. Other consequence categories (such as released amounts of hydrocarbons and chemicals, or enforced clean-up of land and water) can not be compared over all systems since they are either associated with a subset of the analysed systems or the completeness of data differs so much between the systems that a comparison does not appear to be meaningful (but also see chapter 3 and Appendix A on oil spills).

In fact, it needs to be acknowledged that for some of the categories that are compared in this chapter the completeness is quite heterogeneous across the various options. In relative terms the fatality records show the best completeness and are reasonably homogeneous in this respect. Probably the least complete and perhaps the most uncertain information concerns costs of accidents (i.e. economic damages caused by accident, normally excluding costs of health effects). The cost data are not consistent due to the partially uncontrolled differences in the cost definition, coverage (frequently not specified in the original sources) and interpretation (e.g. claimed, settled and real costs). The cost elements that have been included in the various estimates may include different components, which makes the comparison quite unbalanced. Nevertheless, we decided to include also comparisons of economic losses since they reflect the current state of knowledge. The above reservations should, however, be kept in mind when viewing the results.

In contrast to aggregated indicators, frequency-consequence (F/N) curves are only provided for fatalities as these are of primary interest and some of the other indicators are based on too limited evidence to construct meaningful curves.

For comparative purposes, the data were normalized on the basis of the unit of electricity production for the different energy sources. For nuclear and hydro power the normalisation is straight-forward since in both cases the generated product is electrical energy. In the case of coal, oil, natural gas and LPG the thermal energy was converted for the purpose of comparisons showed in this chapter to an equivalent electrical output using a factor of 0.35. It should be noted that for external cost estimates the actual efficiency for the plant of interest may be employed to obtain the case-specific estimates.

Allocation of damages to countries exporting or importing energy carriers were performed for oil, natural gas and LPG chains, as described in Burgherr et al. (to be published). This partial reallocation of damages to OECD countries takes into account imports of the respective energy carriers from non-OECD countries.

The use of Gigawatt-electric-year ($GW_e yr$) was chosen for normalisation because large individual plants have typically capacities of the order of 1 GW of electrical output (GW_e). A series of aggregated results is shown in Table 8 and Figures 11 to 14. Table 8 summarizes the most central results for severe accident fatality rates. Data for the various energy chains are presented on a worldwide basis, as well as for OECD and non-OECD countries, with and without allocation. Nuclear energy is included in the comparisons, based on a limited update of earlier evaluations Hirschberg et al. (1998). Since no nuclear severe accidents occurred since 1998 these estimates are still considered relevant though certainly some refinements could be of interest.

It is important to emphasize the differences in the extent of the statistical material available for the different energy sources. While the historical experience with severe accidents is extensive in the case of fossil energy chains, the statistical evidence available for severe nuclear accidents resulting in fatalities is limited to one accident. Also for hydro power the statistical basis is relatively limited.

Also note that only immediate fatalities are covered here. Latent fatalities, of particular relevance for the Chernobyl accident, are commented on and further addressed within frequency-consequence curves.

Table 8: Experience-based immediate fatality rates associated with severe accidents within full energy chains for the period 1969-2000. Results for OECD and non-OECD countries are given with and without allocation of damages.

Energy chain			Number of fatalities per GW_eyr				
	Number of severe accidents world-wide with fatalities			*No allocation*		*With allocation*	
	# accidents	# fatalities	Worldwide	OECD	Non-OECD	OECD	Non-OECD
Coal (a)	1221 177	25'107 7090	0.876 0.690	0.157	1.605 0.597	0.185 0.163	1.576 0.589
Oil	397	20'283	0.436	0.135	0.897	0.392	0502
Natural Gas	125	1978	0.093	0.080	0.111	0.091	0.096
LPG	105	3921	3.536	1.957	14.896	3.317	5.112
Hydro (b)	11 10	29'938 3938	4.265 0.561	0.003	10.285 1.349	0.003	10.285 1.349
Nuclear	1	31	0.0064	0	0.048	0	0.048

(a) second line: accidents from the Chinese coal chain excluded
(b) second line: Banqiao/Shimantan accident with 26‘000 fatalities excluded

The present work shows that significant differences exist between the aggregated, damage rates assessed for the various energy carriers. However, one should keep in mind that from the absolute point of view the fatality rates are in the case of fossil sources small when compared with the corresponding rates associated with the health impacts of normal operation. For this reason the evaluation focuses here on the relative differences between the various energy carriers.

The broader picture obtained by coverage of full energy chains leads on the world-wide basis to aggregated immediate fatality rates being much higher for the fossil fuels than what one would expect if only power plants were considered. The highest rates apply to LPG and hydro, followed by coal, oil, natural gas and nuclear[3]. In the case of nuclear, the estimated latent fatality rate solely associated with the only severe (in terms of fatalities) nuclear accident (Chernobyl), exceeds all the above mentioned immediate fatality rates. However, in view of the drastic differences in design, operation and emergency procedures, the Chernobyl-specific results are considered not relevant for the "Western World"; in fact the Chernobyl accident is currently due to similar though partially less pronounced reasons hardly representative for the nuclear power plants operating in the non-OECD countries. Given lack of statistical data, results of state-of-the-art Probabilistic Safety Assessments (PSAs) for representative western plants are used as the reference values (but see Hirschberg et al., 1998). PSA-based latent fatality rates for western plants are in the range $10^{-3} – 10^{-1}$ per $GW_e yr$. Delayed fatalities are likely to have occurred for the other chains with no records available; their significance should, however, be incomparably smaller in comparison with the Chernobyl accident.

Figure 11 shows the estimated number of <u>immediate</u> fatalities, injured and evacuees per unit of energy for the period 1969-2000. Results for the different energy chains are given for OECD, non OECD and EU15. Comments on the relative completeness of the data concerning the three damage categories have been stated earlier.

Generally, the immediate failure rates are for all considered energy carriers significantly higher for the non-OECD countries than for OECD countries. In the case of hydro and nuclear the difference is in fact dramatic. The recent experience with hydro in OECD countries points to very low fatality rates, comparable to the representative PSA-based results obtained for nuclear power plants in Switzerland and in USA. The Figure also shows that the Chinese coal chain[4] should be treated separately as its accident fatality rates is about ten times higher than in other non-OECD countries and about forty times higher than in OECD countries

Overall, values for EU15 alone are lower than for OECD countries, but differences are in some cases minimal. However, it should be considered that the statistical basis for EU15 is much smaller, and that OECD includes several countries that have joined only recently. Therefore, integration of accession countries could raise EU15 rates to levels similar for OECD countries.

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[3] Note that the ranking is depending on wether the largest hydro accident at Banqiao/Shimantan with 26'000 fatalities is included or not.

[4] Only data for 1994-1999 are representative because of substantial underreporting in earlier years.

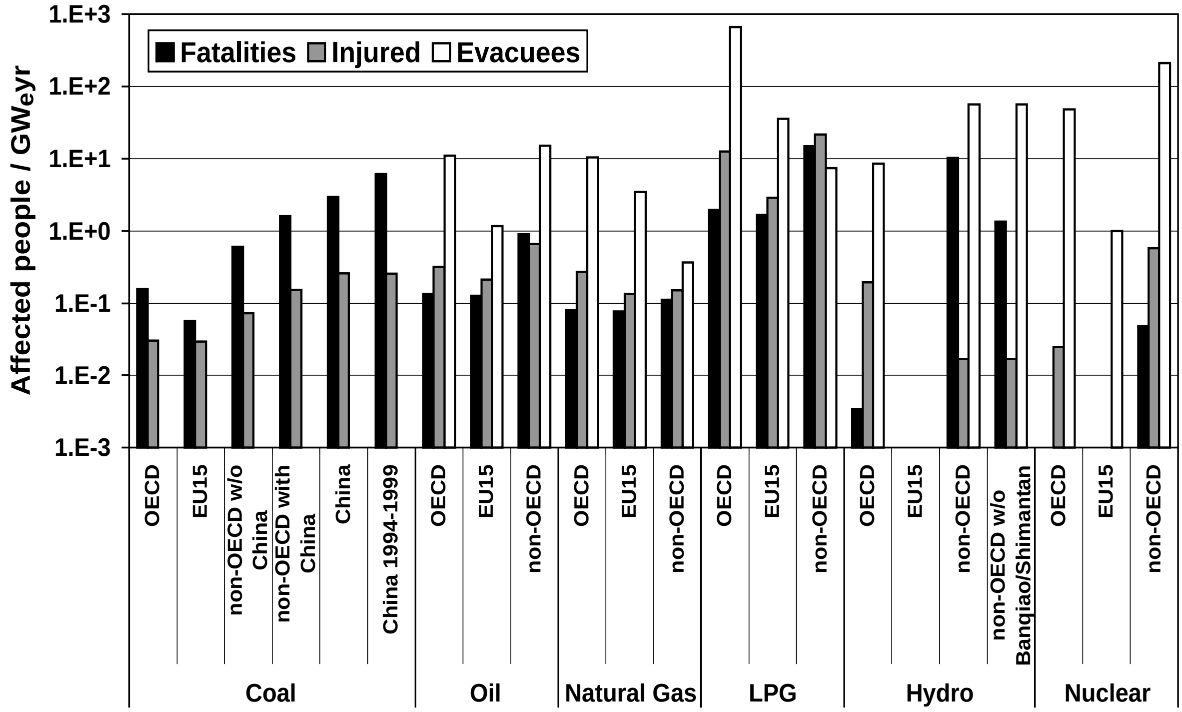

Figure 11: Comparison of aggregated, normalized, energy-related damage rates, based on historical experience of severe accidents that occurred in OECD countries, non-OECD countries and EU15 for the period 1969-2000, except for data from the China Coal Industry Yearbook that were only available for the years 1994-1999. No reallocation of damages between OECD and non-OECD countries was used in this case. Note that only immediate fatalities were considered, but latent fatalities, of particular relevance for the nuclear chain, are commented in the text.

Figure 12 shows the numbers of immediate fatalities, injured and evacuated persons per unit of energy for the period 1969-2000. Results are based on the weighted allocation of damages that occurred in non-OECD countries within fossil energy chains to the corresponding damages in OECD countries.

Severe fatality rates for the oil chain exhibited the most distinct increase for OECD countries and decrease for non-OECD countries in comparison to the rates without allocation, as shown before. Changes for LPG were still substantial but less pronounced, and distinctly smaller for the natural gas and coal chains. Relative rankings for other indicators were the same, but differences were smaller.

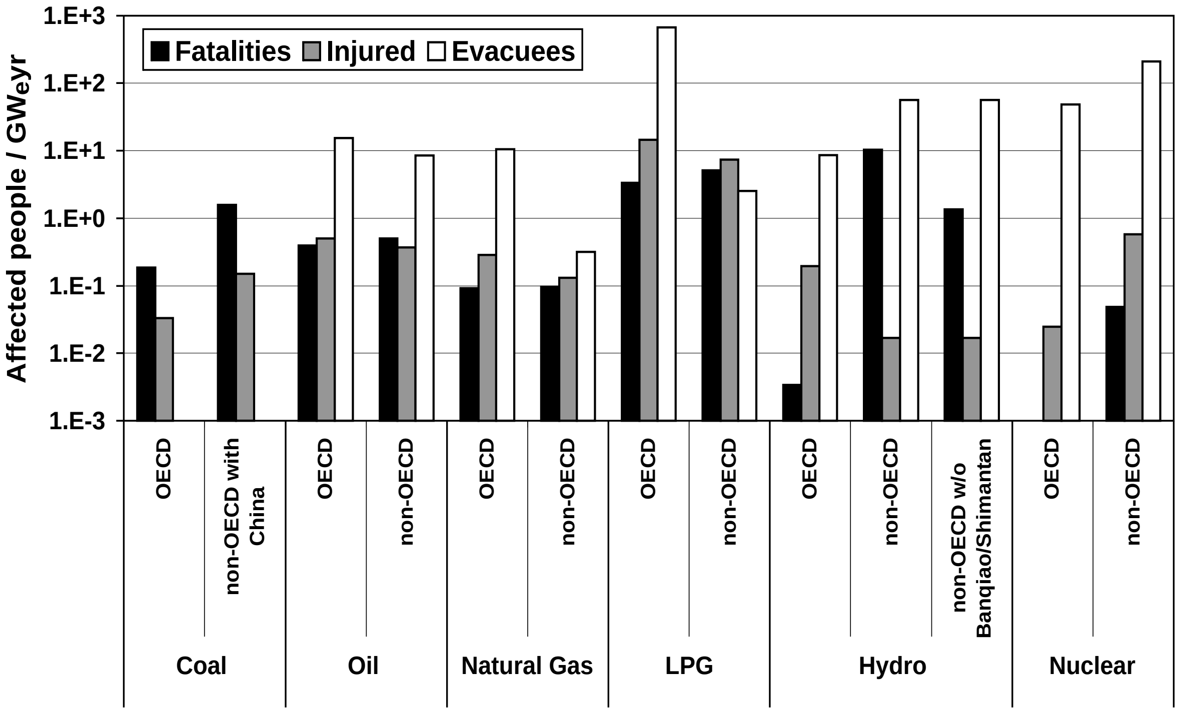

Figure 12: Comparison of aggregated, normalized, energy-related damage rates, based on historical experience of severe accidents that occurred in OECD and non-OECD countries for the period 1969-2000. Damage indicators per unit of energy were estimated on reallocation of damages to OECD countries taking into account imports of fossil energy carriers from non-OECD countries. Note that only <u>immediate</u> fatalities were considered, but <u>latent</u> fatalities, of particular relevance for the nuclear chain, are commented in the text.

The comparison of economic damages is limited by incompleteness and some serious inconsistencies. First, the estimates of monetary losses are not available for a major part of non-nuclear accidents. Second, the cost elements covered, i.e., the boundaries of the calculation, are normally not documented and may vary widely from case to case. Third, the nature of the reported costs may be different - there is normally a large discrepancy between the compensation paid by insurance companies, claimed damages, real damages, direct costs and indirect costs. In the nuclear case the costs of two accidents have been included, namely TMI and Chernobyl. They are dominated by the latter accident with more than one order of magnitude discrepancy between the lower and higher bound of this estimate.

In Figure 13 (no allocation) and Figure 14 (full allocation for fossil energy chains) the distinction is made between OECD and non-OECD countries, respectively. For comparison, estimates without allocation include EU15 and separate values for the Chinese coal chain.

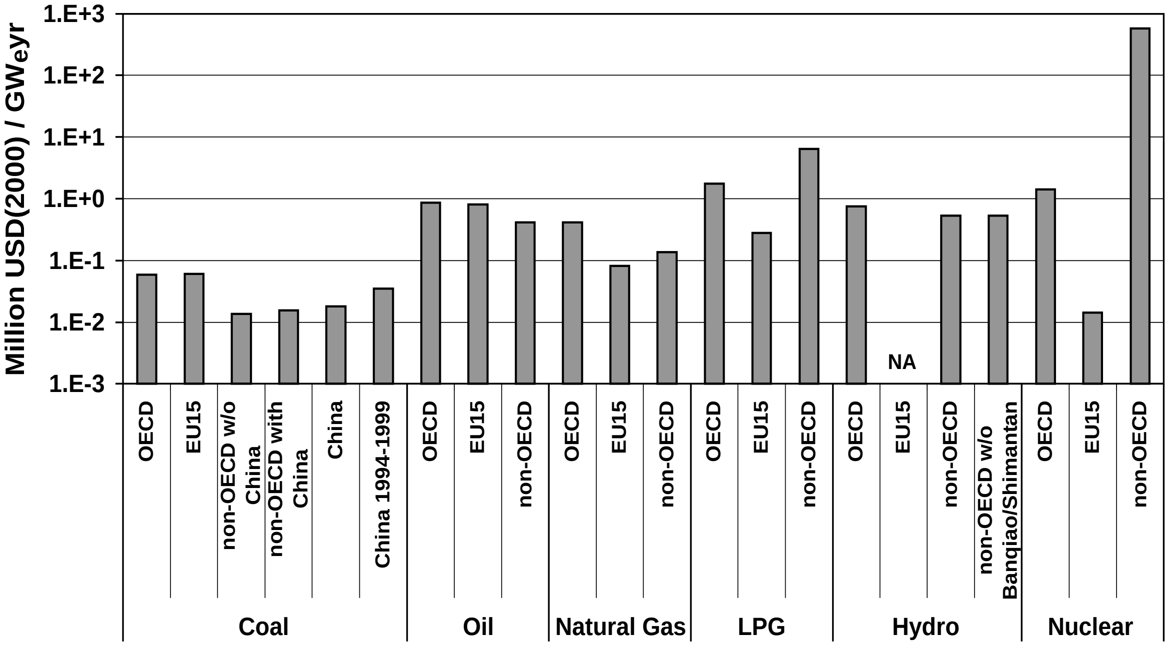

Figure 13: Comparison of aggregated, normalized, energy-related damage rates, based on historical experience of severe accidents that occurred in OECD countries, non-OECD countries and EU15 for the period 1969-2000, except for data from the China Coal Industry Yearbook that were only available for the years 1994-1999. No reallocation of damages between OECD and non-OECD countries was used in this case.

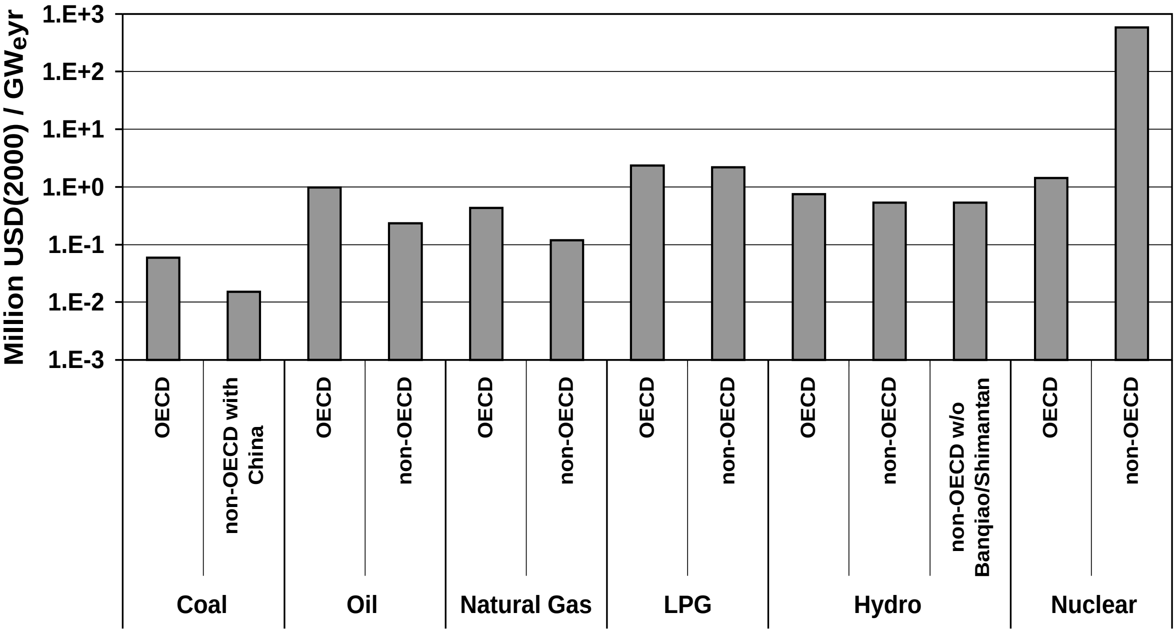

Figure 14: Comparison of aggregated, normalized, energy-related damage rates, based on historical experience of severe accidents that occurred in OECD countries and non-OECD countries for the period 1969-2000. Results are based on reallocation of damages to OECD countries taking into account imports of fossil energy carriers from non-OECD countries.

The results obtained for economic losses and their interpretation are subject to the serious reservations mentioned above. Due to the devastating damages associated with the Chernobyl accident the normalised monetary damages are clearly highest for the nuclear chain, followed by LPG, oil, hydro, natural gas and coal. Consideration of the regional distribution of accidents leads to a somewhat different ranking for the most developed countries. It is also worthwhile to note that the partially artificial limitation of the evaluation period strongly influences the results. For example, according to the records some of the hydro accidents that occurred further back in time resulted in extremely high damages.

The comparison of results is not limited to the aggregated values obtained for specific energy chains. Also frequency-consequence curves are provided. They reflect implicitly the above ranking but provide also such information as the observed or predicted chain-specific maximum extents of damages. This perspective on severe accidents may lead to different system rankings, depending on the individual risk aversion.

Figure 15 shows the frequency-consequence curves for OECD countries. Among the fossil chains natural gas has the lowest frequency and LPG the highest frequency of severe accidents involving fatalities, whereas coal and oil chains are ranked inbetween. Hydro experience in OECD countries is significantly lower than for fossil chains, but with respect to fatalities there is only one severe accident for the evaluation period considered. Finally, expectation values for severe accident fatality rates associated with hypothetical nuclear accidents are lowest among the relevant energy chains.

Figure 16 compares frequency-consequence curves for non-OECD countries. Fossil energy chains in non-OECD countries display a similar ranking as for OECD countries, except for the Chinese coal chain that exhibits significantly higher accident frequencies than in other non-OECD countries. However, the vast majority of severe coal accidents in China results in less than 100 fatalities. Accident frequencies of the oil and hydro chains are also much lower than for the (Chinese) coal chain, but maximum numbers of fatalities within the oil and hydro chains are one respectively two orders of magnitude higher than for coal and natural gas chains. Finally, expectation values for severe accident fatality rates associated with the nuclear chain (Chernobyl) are relatively low, but the maximum credible consequences may be very large, i.e. comparable to the Banqiao and Shimantan dam accident that occurred in China in 1975.

However, the large differences between Chernobyl-based estimates (Figure 16) and probabilistic plant-specific estimates for Mühleberg (Figure 15) illustrate the limitiations in applicability of past accident data to cases which are radically different in terms of technology and operational environment.

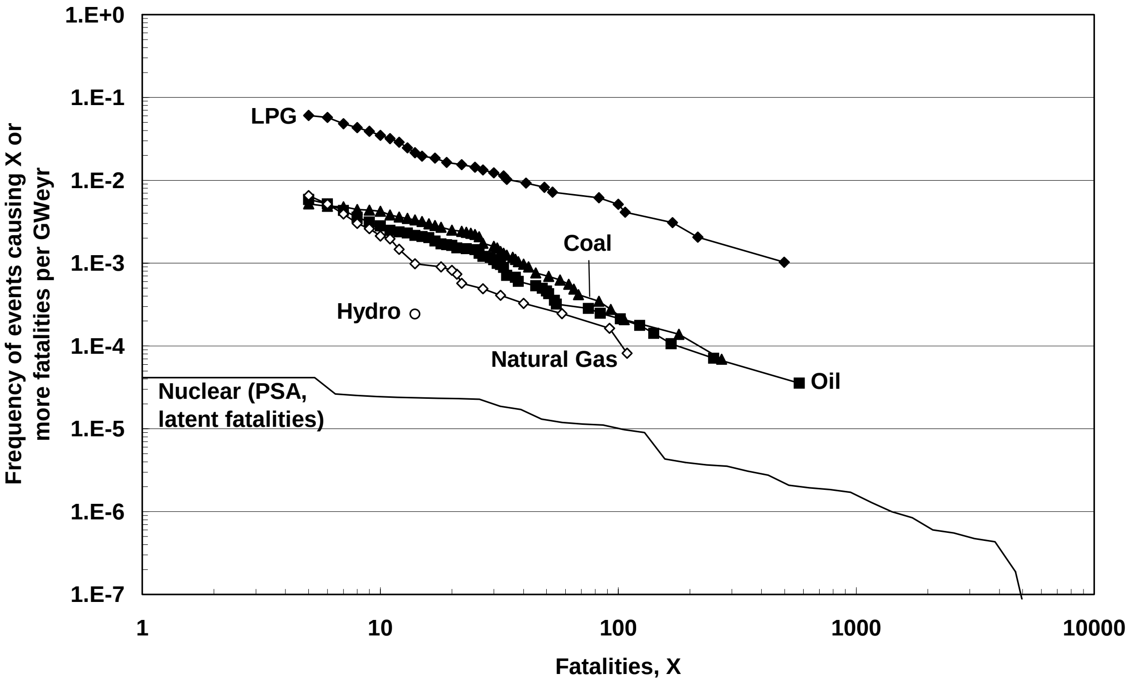

Figure 15: Comparison of frequency-consequence curves for full energy chains in OECD countries for the period 1969-2000. The curves for coal, oil, natural gas, LPG and Hydro are based on historical accidents and show immediate fatalities. For the nuclear chain, the results originate from the plant-specific Probabilistic Safety Assessment (PSA) for the Swiss nuclear power plant Mühleberg and reflect latent fatalities.

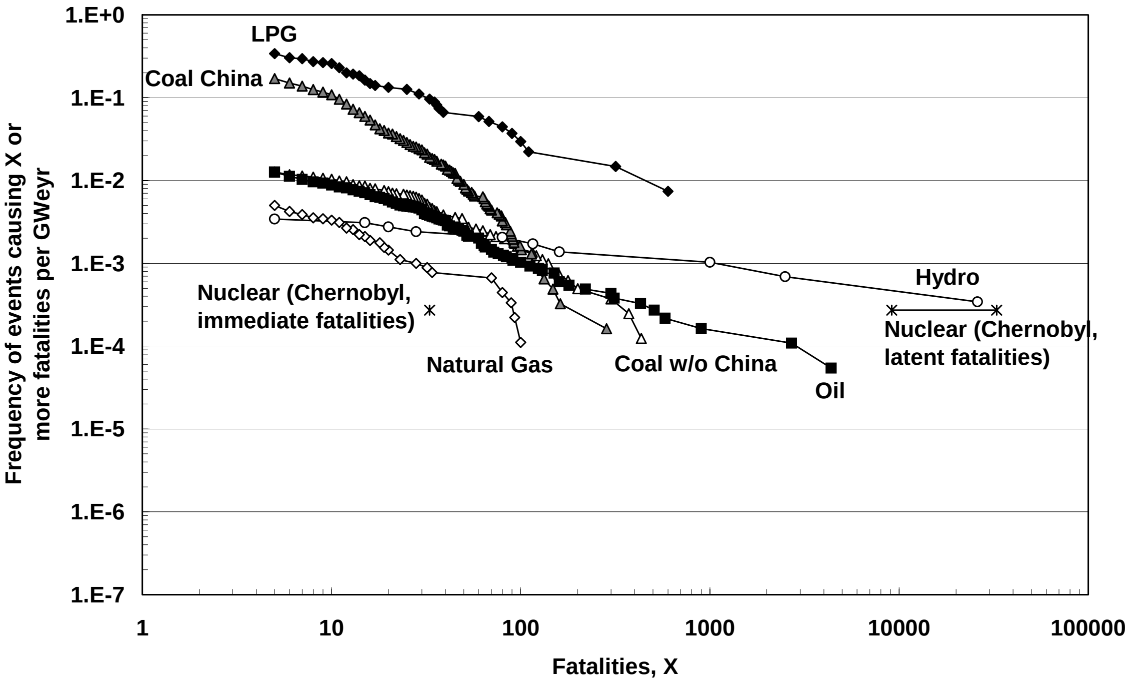

Figure 16: Comparison of frequency-consequence curves for full energy chains in non-OECD countries for the period 1969-2000. The curves for coal w/o China, coal China, oil, natural gas, LPG and Hydro are based on historical accidents and show immediate fatalities. For the nuclear chain, the immediate fatalities are represented by one point (Chernobyl); for the estimated Chernobyl-specific latent fatalities lower and upper bound are given.

Figure 17 gives frequency-consequence curves for EU15. Generally, ranking of curves for EU15 experience is similar to OECD countries, although some minor deviations exist. For example, the coal chain has the lowest accident frequency below a threshold of about 12 fatalities; above which then natural gas has the best performance. Maximum numbers of fatalities for EU15 are smaller for all fossil chains, when compared with OECD countries. Concerning hydropower, no accident with at least 5 fatalities occurred in EU15 during the period of evaluation. Overall, it appears that OECD experience may serve as a robust estimate for EU15, particularly in view of the relatively small historical accident database that is available for EU15.

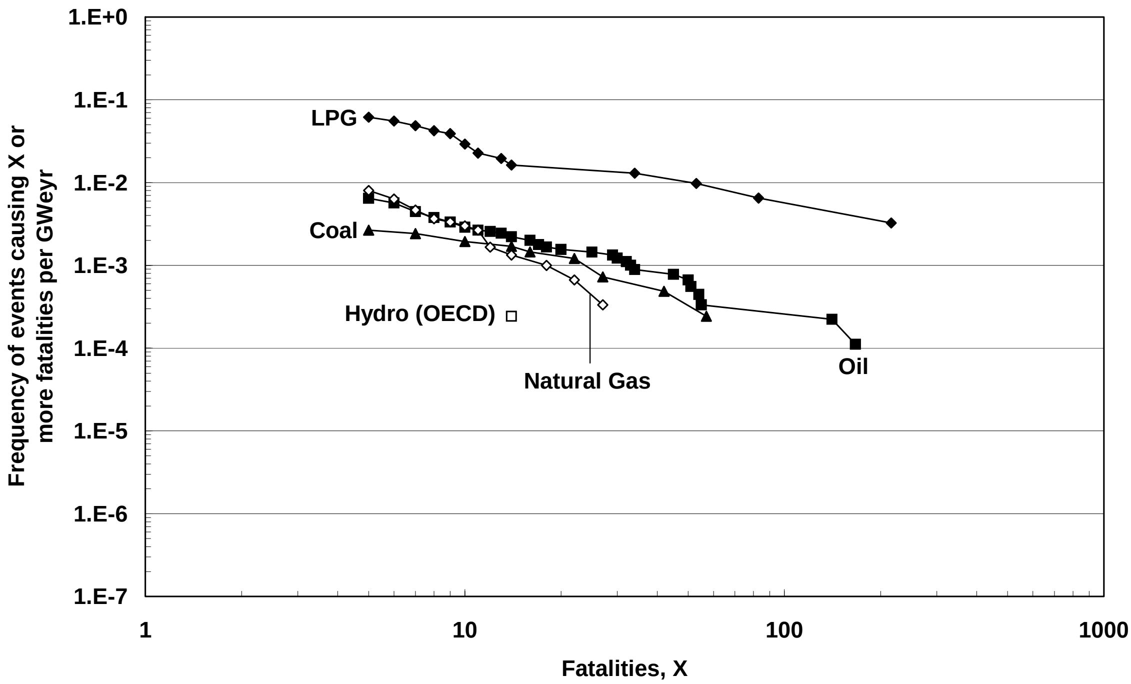

Figure 17: Comparison of frequency-consequence curves for full energy chains EU15 for the period 1969-2000. The curves for the different energy chains are based on historical accidents and show <u>immediate</u> fatalities. Hydropower data represent OECD experience because no dam failure with at least 5 fatalities occurred in EU15 during 1969-2000.

As with aggregated indicators, reallocation of accidents was carried out to obtain the respective frequency-consequence curves for OECD (Figure 18) and non-OECD countries (Figure 19). This did not change rankings of energy chains, but affected relative differences, i.e., differences between coal and oil chains became smaller for OECD countries but increased for non-OECD countries as a consequence of the allocation procedure.

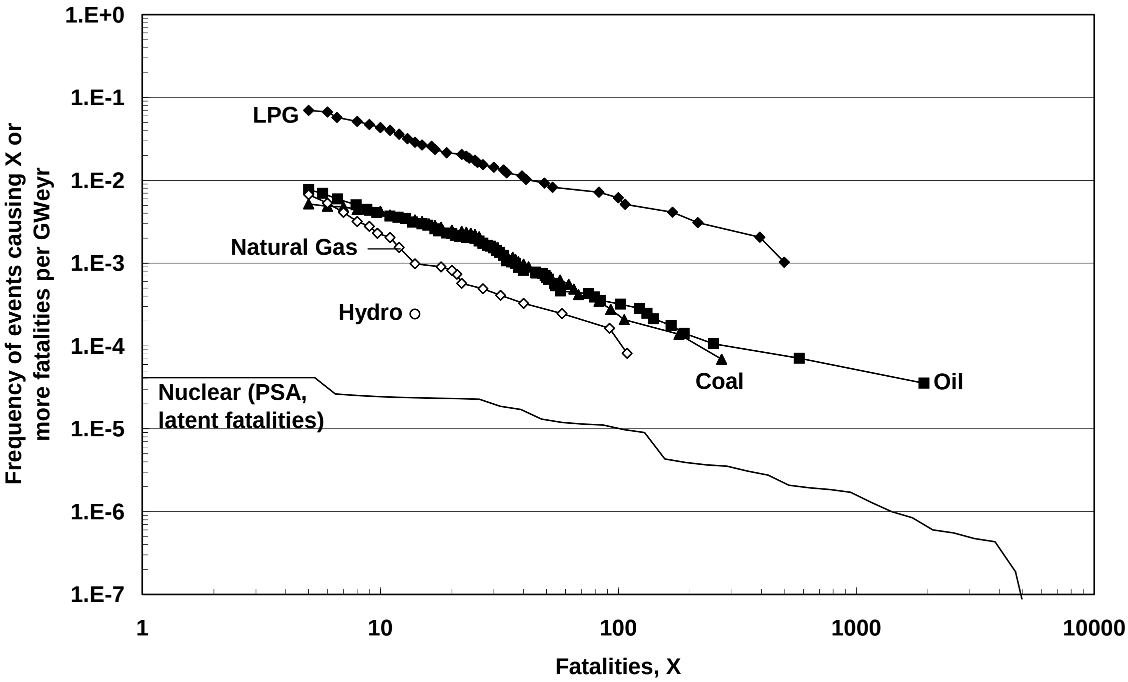

Figure 18: Comparison of frequency-consequence curves for full energy chains in OECD countries with full reallocation for the period 1969-2000. The curves for coal, oil, natural gas, LPG and Hydro are based on historical accidents and show <u>immediate</u> fatalities. For the nuclear chain, the results originate from the plant-specific Probabilistic Safety Assessment (PSA) for the Swiss nuclear power plant Mühleberg and reflect <u>latent</u> fatalities.

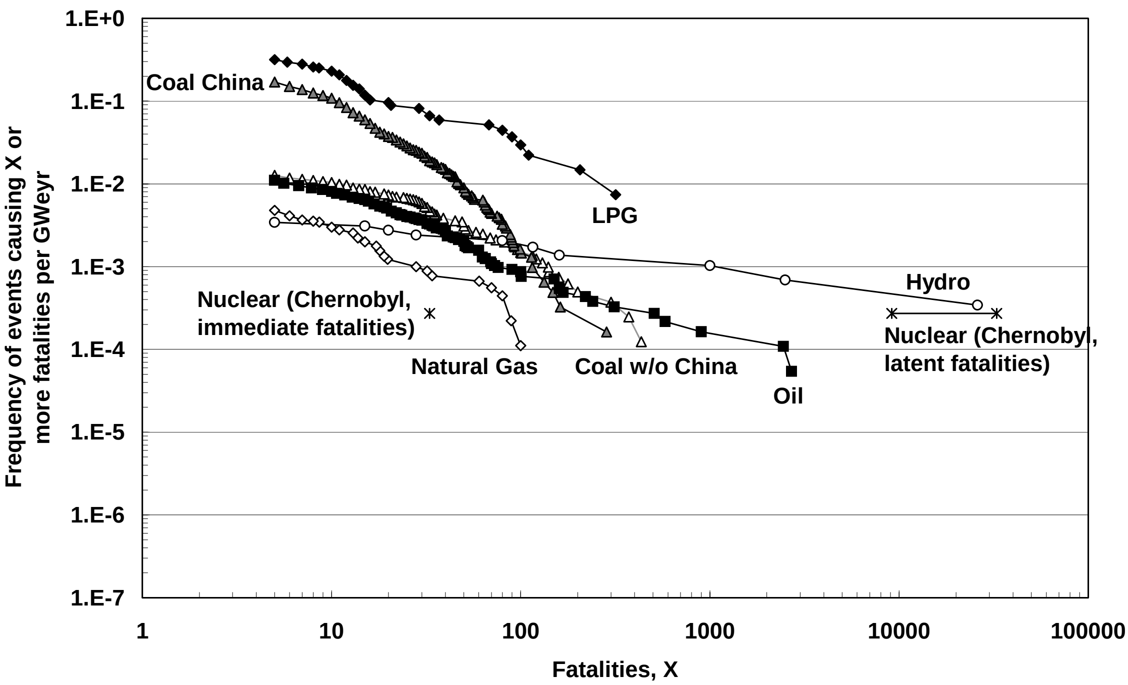

Figure 19: Comparison of frequency-consequence curves for full energy chains in non-OECD countries with full reallocation for the period 1969-2000. The curves for coal w/o China, coal China, oil, natural gas, LPG and Hydro are based on historical accidents and show <u>immediate</u> fatalities. For the nuclear chain, the <u>immediate</u> fatalities are represented by one point (Chernobyl); for the estimated Chernobyl-specific <u>latent</u> fatalities lower and upper bound are given.

7. Econometric valuation of Severe accidents

7.1 Unit values for impact categories

In the following subsections we summarize the possibilities for deriving appropriate unit values for each of these impacts. Our conclusions are drawn from the findings of a literature review that we have undertaken.

Calculations for total damages and external costs of severe accidents for the different energy chains were based on these unit values. Results are presented in chapter 7.2.

7.1.1 Components of external costs associated with health impacts

There is an established methodology - adopted in ExternE and related projects - for estimating the valuation of health risks. This involves - as the starting point for the valuation of health end-points and a number of the other impact categories considered below - the identification of the components of changes in welfare. These components should be summed to give the total welfare change, assuming no overlap between categories. The three components include:

– *Resource costs* - medical costs paid by the health service in a given country or covered by insurance, and any other personal out-of-pocket expenses made by the individual (or family).

– *Opportunity costs* - the cost in terms of lost productivity (work time loss (or performing at less than full capacity)) and the opportunity cost of leisure (leisure time loss) including non-paid work.

– *Dis-utility* - other social and economic costs including any restrictions on or reduced enjoyment of desired leisure activities, discomfort or inconvenience (pain or suffering), anxiety about the future, and concern and inconvenience to family members and others.

We discuss the potential impact categories listed above with these components of WTP/WTA in mind.

7.1.2 Premature mortality

7.1.2.1 Conceptual background

The value of a statistical life (VSL) is a convenient figure for evaluating policies that reduce risk of death and is the total willingness to pay for the policy that is predicted on an ex ante basis to result in an one additional/less death in the population. It can also be defined as the aggregate willingness to pay for a measure saving a number of lives divided by the number of lives saved. Alternatively, a VSL is derived from the marginal rates of substitution between income and risk of death of groups of affected individuals. The purpose of estimating the VSL is to provide some welfare basis for policy making involving social decisions when premature human deaths are to be considered.

As an example suppose that the average amount a group of individuals is each willing to pay €2 to reduce risk of death in one year by 1:1’000’000. Then the VSL is estimated as:

€2 x 1’000’000 = €2 million

The Willingness to Pay (WTP) approach for identifying the VSL has its basis in the assumption that changes in individuals’ economic welfare can be valued according to what they are willing (and able) to pay to achieve that change. According to this assumption, individuals treat longevity like any other consumption good and reveal their preferences through the choices that involve changes in the risk of death and other economic goods whose values can be measured in monetary terms. That is, in many situations individuals act as if their preference functions included life expectancy or the probability of death as arguments, and make a variety of choices that involve trading off changes in their risk of death for other economic goods. When what is being changed can be measured in monetary terms, the individual willingness to pay is revealed by these choices. This WTP is the basis of the economic value of reductions in the risk of death.

In the health economics literature, various methods for empirical estimation of willingness to pay measures have been utilised, each providing a method for deriving measures for individuals making trade-offs between risks to life and health and other consumption goods and services. These methods include the Compensating or Hedonic Wage, the Contingent Valuation, the Hedonic Property Value, and the Averting Behaviour methods. Table 9 gives examples of VSL estimates based on labour market studies in Europe.

7.1.2.2 VSL Measures in the energy supply accidents context

As described in previous ExternE documents (e.g., European Commission, 1995), estimation of the value of a lost life or of a prevented fatality (VPF) is fraught with conceptual and empirical difficulties associated with the fact that there is no direct market for values to be reflected in[5]. Two issues should be highlighted in relation to our present needs. First, estimates of the VSL that have been made to date have primarily been derived in the context of road traffic or workplace accidents. None is known to have been estimated in the context of energy supply operations and this therefore raises a question about the appropriateness of transfer between contexts. The second issue is that in order to identify a unit value for the risk of premature death in the energy supply context we need

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5 A detailed discussion is provided in Chapter III (Monetary valuation of increased mortality from air pollution) of this report.

7 Details and discussion on the uncertainties regarding the estimates are provided in Chapter III.

to consider whether or not – and to what degree – the WTP is measuring an external cost. For instance, if an employee who is working in the energy supply industry is fully compensated through the wage rate for the risk of a fatal accident to which he is exposed then the cost is fully internalised in existing financial flows. These two issues are discussed at some length below and we find it convenient to consider WTP for mortality risks to employees and the general public separately.

7.1.2.3 Work-Related Accidents

The derivation of a unit value for this impact is presented in two stages. First, we identify a VSL unit value, before estimating the extent to which the value is internalised in existing financial flows.

The hedonic wage method would seem to be the appropriate approach to empirically estimate work-related values of a statistical life, since it uses the wage-risk trade-offs (and other factors that affect wages) to estimate wage differentials related to different mortality risks. However, there are a number of difficulties associated with the estimation of VSLs using this method. Principal amongst these difficulties – based upon a review paper by (Viscusi & Aldy, 2003) are the following:

- Risk data: the standard approach in the literature is to use industry-specific or occupation-specific risk measures reflecting an average of several years of observations for fatalities, which tend to be rare events. However, the choice of the measure of fatality risk can significantly influence the magnitude of the risk premium estimated through regression analysis.
- Omitted variables bias and endogeneity: failing to capture all of the determinants of a worker’s wage in a hedonic wage equation may result in biased results if the unobserved variables are correlated with the observed variables, since dangerous jobs are often unpleasant in other respects. For example, one may find a correlation between injury risk and physical exertion required for a job or risk and environmental factors such as noise, heat, or odour. Various studies have demonstrated how omitting injury risk affects the estimation of mortality risk, indicating that a positive bias in the mortality risk measure is introduced when the wage equation omits injury risk.
- While including injury risk in a regression model could address concern about one omitted variable, other possible influences on wages that could be correlated with mortality risk may not be easily measured. For example, individuals may systematically differ in unobserved characteristics, which affect their productivity and earnings in dangerous jobs, and so these unobservable will affect their choice of job risk (Garen, 1988, 1998). The studies reviewed by Viscusi & Aldy (2003) indicate that models that fail to account for heterogeneity in unobserved productivity may bias estimates of the risk premium by about 50%.

- Endogeneity: the issue here being that the dependent variable (wage) is explained by, among others, the risk variable, which simultaneously depends on wage, since "the level of risk that workers will be willing to undertake is negatively related to their wealth, assuming that safety is a normal good." (Viscusi, 1978). Gunderson & Hyatt (2001) empirically tested the alternative econometric models suggested by Viscusi (1978) and Garen (1988), identifying significant differences in the VSL estimates between the usual econometric model (OLS) and the proposed alternatives (€ 2.8 million to € 12.8 million).

These difficulties with the reliability of the estimation methods are exacerbated when we try to identify a typical average unit value by the wide range of values that result from the wage-risk studies. A sample of the studies undertaken in the EU, presented in Table 9 below, demonstrate this.

Table 9: Summary of European Labour Market Studies of the VSL.

Author (year)	Country	Mean risk	Implicit VSL (€ million, 2000 prices)
Martin and Psacharopoulos (1982)	UK	0.0001	4.2
Weiss, Maier & Gerking (1986)	Austria	n.a.	3.9 – 6.5
Siebert & Wei (1994)	UK	0.000038	9.4 – 11.5
Sandy & Elliot (1996)	UK	0.000045	5.2 – 69.4
Arabsheibani & Martin (2000)	UK	0.00005	19.9
Sandy, Elliot, Siebert & Wei (2001)	UK	0.000038	5.7 – 74.1

As a consequence of the issues raised above we do not find these estimates highly robust. The alternative source of a unit value for a VSL is to use a value derived from other valuation methods. As part of this NewExt project, a contingent valuation study was conducted in order to estimate the willingness to pay to reduce risks of death in three European countries[7]. The contingent valuation survey considered a context-free scenario where the respondent faced two different reductions in his or her probability of death. Because of this context-free characteristic, the results can be extrapolated to different situations involving risks of death, as in the context of accidents in non-nuclear fuel chains.

Table 10 summarises the results[8]. It presents values for VSL derived from the WTP for a 5 in 1000 reduction in mortality risk, and additionally gives equivalent values for the Value of Life Years Lost (VLYL).

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8 The studies also considered future risk reductions, which are more appropriate in contexts involving latency periods between exposure and death, like in the context of air pollution. In the context of accidents in non-nuclear chains, immediate risk reductions are appropriate.

Table 10: Value of Statistical Life Estimates.

	Median	Mean
VSL (€)	1’044’154	2’153’454
VLYL (€)	47’640	98’251

We recommend the use of median values because the econometric analysis suggests that whilst median values from various assumed distributions agree, the same does not hold for mean WTP. We regard median WTP as a conservative, but robust and more reliable, estimate. As approximate rounded numbers we suggest that the values presented in Table 10 could be €1 million for VSL and €50’000 for VLYL.

Fatalities that occur to employees involved in fuel cycles may already be at least partly internalised in producer costs, either through ex ante wages that account for fatality risks or through ex post compensation to families of the victim. Internalisation of the risk of fatality is likely to the extent that workers can be assumed to be well informed about the risks that they actually face in their work and that the part of the labour market to which these risks apply is competitive and flexible. Evidence of the validity of these assumptions hold is not easy to come by. In order to identify the degree to which internalisation of mortality and morbidity risks exists in the energy supply sector we would ideally need to have a quantitative estimate of the extent to which actual wage rates differ from what they would be in a perfect market, within this sector. There is no evidence from wage simulation models of this measure and results in this regard from wage-risk studies, (the explanatory power of the risk variable), vary enormously.

In the absence of direct evidence of the degree of internalisation that we can assume, we have investigated the possibility of using a proxy for the degree to which workers are well informed of mortality and morbidity, and are able to express this in wage negotiation and settlement. To this end, we have looked at the importance of education and unionisation as explanatory variables. Dorman & Hagstrom (1998) finds that whilst wage levels increase with the education level of labour force there is no robust way in which these results can be related to differing levels of mortality/morbidity risk. Evidence regarding the role of unions (e.g. reported in CSERGE et al., 1999) in determining the level of risk premiums is also not particularly convincing since whilst some studies found union affiliation had an insignificant impact on risk premium, others found that higher union risk premiums existed.

Given the lack of any satisfactory measure of internalisation, we are obliged to rely on judgement. On this basis we would suggest using 80% as a direct proxy value for the central degree of internalisation that may be assumed in OECD countries. High and low ranges of internalisation may reasonably be assumed to be 100% and 70% respectively, reflecting the fact that in industrialized economies occupational risk is recognised as being substantially internalised. For non-OECD countries we recognise that whilst some economies, e.g. in Eastern Europe, are less effective and that a lower degree of internalisation is to be expected, others, e.g. in East Asia, are much more market-orientated and are better able to reflect risk premiums according to the preferences of market participants. In the absence of hard data we suggest that a wide range of 0% to 100% internalisation, with a central value of 50%, is not too unreasonable to assume. It should be emphasised that the lack of data with which to validate these percentages significantly limits the extent to which they can be regarded as reliable.

7.1.2.4 Non-Work-Related Accidents

In addition to work-related accidents, some fuel chain accidents affect a great number of people not related to the production per se, the general public. For example, floods generated from hydro-dam collapses may affect residents downstream of the dam. Two issues are important when considering valuation of risks of non-work related accidents in non-nuclear fuel chains: the fact that these risks are involuntarily taken by the population affected by accidents, and that the choices that individuals are able to make to allocate the perceived risks of potential accidents in fuel chains determine the degree to which the costs are internalised. These issues are considered in more depth below before making recommendations for final unit values.

Involuntariness

The degree of involuntariness, or the lack of personal choice on the exposure to risks, may differ between different accident contexts. The argument here is that whereas road accidents are more or less voluntary to the extent that the risk is in the individual's control and has responsibility for his/her actions, the degree of voluntariness can be judged to be very low for both employees and the general public who suffer fatalities from accidents in the fuel cycle. Evidence is sparse but one study (Jones-Lee & Loomes, 1995) identifies a 50% premium between the event of an underground train accident (involuntary) and road accident (voluntary), which did not appear to be the result of any particular additional dread of underground accidents relative to road accidents. It is proposed that this premium be adopted in sensitivity analysis.

Internalisation

Kunreuther (2001) argued that individuals can take two actions to reduce their losses from natural disasters and accidents and so internalise the risk: up-front expenditures to avoid or mitigate losses which provide benefits over the life, or the purchase of insurance which provides the policyholder with financial protection against a disaster loss for a fixed period of time in return for a premium to the insurance company. In determining which actions can be taken to reduce their losses from accidents an individual would need to consider: the probability that the event (accident) will occur; the resulting loss associated with the event, and; the cost associated with protection that reduces this loss from an accident. Normative models of choice predict that individuals, depending on their aversions to risk, maximise their utilities by choosing between the two different protective measures, buying insurance or mitigation measures.

However, the empirical literature suggests that individuals and firms do not obtain the relevant data or do not undertake the (expected) utility maximising problem implied by normative models of choice. The factors that lead people to behave differently from what is predicted in normative models of choice are identified (Kunreuther, 2001), as:

- Misperception of the risks – sometimes the probability of occurrence of certain event is overestimated because of media coverage. For example, empirical tests suggest that the likelihood of deaths from widely reported disasters are perceived to be higher than those from events such as diabetes and breast cancer that are not reported in the media in the same way. Past experience may also play an important role in influencing individuals' perception of the probability of occurrence of an event. Individuals tend to perceive that an accident is more likely to occur after experiencing an accident than before the occurrence.

- Low probability events are perceived as impossible events – individuals tend to behave as if they consider the probability of the event occurring to be equal to zero, taking no mitigating measures nor acquiring insurance.

- High discount rates – regarding investment in mitigating measures where the benefits are accrued over time, individuals may have a very high discount rate so that the future benefits are not given much weight when evaluating the protective measure.

- Imperfect capital markets – individuals may not have access to efficient capital markets and therefore may not be able to make a utility-maximising trade-off between accident risk and protection/compensation.

- Role of emotions – judgements on risks are based on dimensions other than probability and monetary losses, such as fear and dread, which have shown to be very critical to individuals' risk perception. With regard to protective behaviour, studies found that people often buy warranties because they want to have peace of mind or reduce their anxiety. In addition, presenting information to individuals in different ways may alter their perception of the risk.

- Ambiguity – or vagueness about the probabilities of losses related to given risks is an attribute that is ignored in normative models of choice, such as expected utility theory, which seems to affect choices individuals make. Empirical tests suggest that ambiguity in risks such as environmental pollution and earthquake losses does make a difference in individuals' willingness to pay to protect them against a risk.

As a consequence of this analysis Kunreuther (2001) concludes that policies for dealing with low-probability-high-consequence events must consider a set of behavioural and capital market factors that are not considered in standard normative models of choice. It is also the case that insurance premiums in general cover only the material losses from e.g. loss of income, and not the costs imposed by pain, suffering, and trauma.

With these issues in mind, we have reviewed the level of insurance compensation payments that are made in the EU. Ex post evidence (Munich Re, 2000) suggests that liability insurers pay a mixture of lump sum and annuities related to wage losses and medical costs for injuries (though in France the indexation is borne by the state) and a mixture of lump sum and annuities related to wage losses to family for fatalities. Coverage for accidents varies over countries and industries but on average between 70% and 80% of material losses are paid i.e. internalised. We therefore assume that 75% of material losses are paid. In order to account for pain and suffering not included in standard compensation payments we make a conservative assumption that this component is equal to 50% of the value of the true material losses. Thus, with a compensation payment made of € 500’000 these assumptions imply a full material cost of € 666’666 (1/0.75 * 500’000) and a full WTP value of € 1 million (adding in 50% of 666’666 €), showing that the compensation payment made is 50% of full internalisation for OECD countries. We adopt this as a central value, with a range of between 30% and 70%. For non-OECD countries, we suggest that a range of between 0% and 50%, with a central value of 20% would be reasonable. Again, the evidence to support these ranges is weak but based purely on the knowledge that many of these countries are characterised as having imperfectly functioning market economies.

While this approach allows for the internalisation of some of the risk, we should note that the component that is internalised is also of interest to policy-makers. It reflects the shifting of the costs of using a resource from the producer of energy to the general public. Hence we recommend that the internalised values also be reported alongside the externalities.

7.1.2.5 Final Remarks on the Value of a Statistical Life

There are three further factors that have been hypothesised as influencing the individual's valuation of a risk of death from fuel-cycle related accidents. We discuss these in the following paragraphs before providing a summary table of recommended values.

<u>The scale of the accident</u>

It has been hypothesised (Savage, 1993) that the scale of an accident (in terms of number of fatalities resulting) may influence the WTP valuation of accident fatalities i.e. that risks of large-scale accidents may be valued more highly. There is to date little evidence available to test this hypothesis. However, a study by (Jones-Lee & Loomes, 1995), compares the valuations that arise out of WTP for large-scale Underground train accidents and third party accidents from proximity to airports with those from small-scale road transport accident. They found no evidence of a significant scale premium, apparently reflecting in part, people's doubts about the preventability of rare, large-scale accidents and the consequent reservations concerning the effectiveness of expenditure aimed at their prevention.

Non-linearity of the size of risk (probability of accident).

It has been noted in earlier ExternE projects that the probability range over which the valuation of mortality risk has been undertaken in road accident studies is typically 10^{-1} to 10^{-5}, whereas the probability of death from accidents may be more likely to be of the order of 10^{-6}. Furthermore, it has been suggested (Lindberg, 1999) that values of mortality risk vary in a non-linear way(Figure 20).

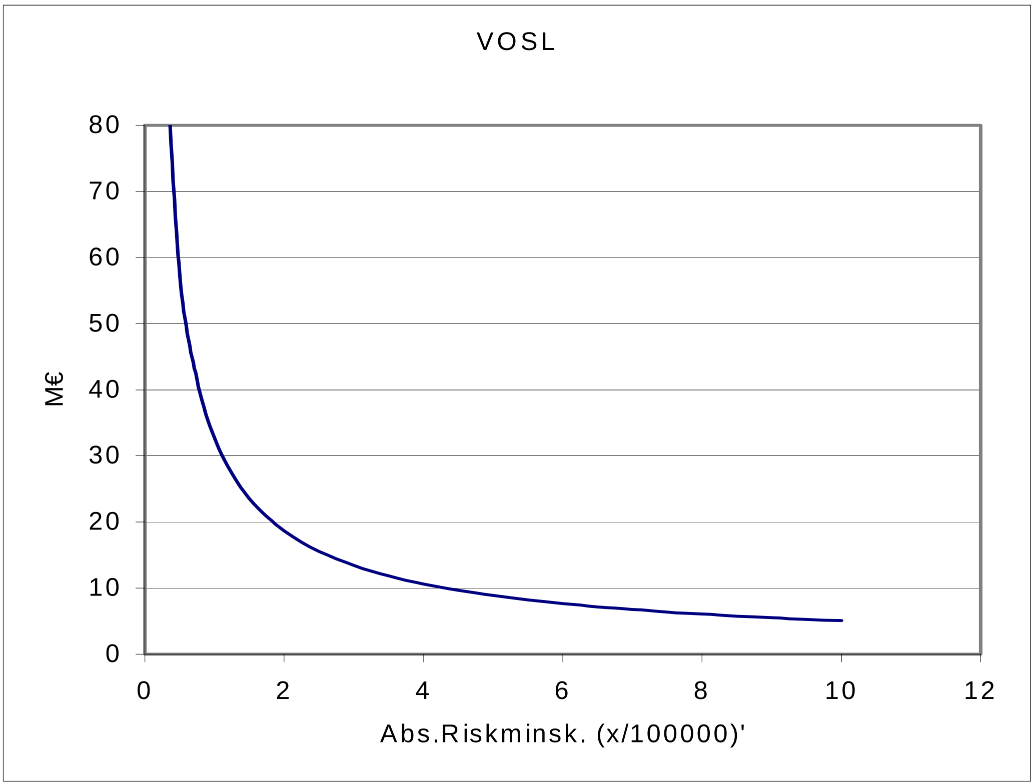

Figure 20: Willingness to Pay for different risk reductions.

As noted in earlier ExternE reports the evidence is not currently sufficient to make any firm proposals on such an adjustment at present.

The age of the victim, and associated life years lost

There appears to be little reason to expect that the average age of the severe accident victim will differ from the average age assumed in the road accident valuation studies (40) and therefore average life expectancy (37.8 years).

Spatial transfer of unit values

Given that incidents of mortality from the non-nuclear fuel cycle occur globally there remains a question as to the appropriate basis for transferring values between EU countries and outside the EU. We recommend that for EU countries themselves there should be no differentiation between individual countries and that common EU values should be utilised. For mortality incidents that occur outside of the EU, economic theory suggests that from an efficiency perspective – if income is assumed to be the principal variable in explaining cross-region variation - that the values could be disaggregated on the basis of local resource costs. In practice, this is measured by purchasing power parity (PPP), and this ratio – referenced to the EU15 – is what we recommend to use here[9]. The PPP ratio should be used for individual countries in future policy analysis. However, where the spread of countries impacted is not known we recommend the use of the unadjusted EU unit value.

In Table 11 we present the unit values that should be used in quantification of accident-related mortality impacts in OECD and non-OECD countries. We assume that the central non-OECD country estimates are representative of industrialised countries of similar per capita income levels to those prevailing in the EU. The minimum and maximum ranges reflect the considerable uncertainty that remains in the derivation of these values. It is recommended that these ranges be used in all quantification of mortality impacts in policy analysis. In addition, the 50% premium of involuntariness exposure to risk noted above should be included in further sensitiveness analysis.

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9 Transfer functions sometimes consider differentials in income elasticities between countries or regions. For a detailed discussion, refer to Markandya (1998).

Table 11: Summary of unit values for occupational and public fatalities in fuel cycle accidents (in €2002), provided for various levels of internalisation (expressed in the last column).

	Central	**Minimum**	**Maximum**	**Proportion of internalisation**
Value of a Statistical Life	1'000'000	400'000	3'310'000	
Occupational fatalities				
Central OECD	200'000	80'000	662'000	0.8
Lower internalisation OECD	300'000	120'000	993'000	0.7
Upper internalisation OECD	0	0	0	1.0
Central Non-OECD	500'000	200'000	1'655'000	0.5
Lower internalisation Non-OECD	1'000'000	400'000	3'310'000	0.0
Upper internalisation Non-OECD	0	0	0	1.0
Public fatalities				
Central OECD	500'000	200'000	1'655'000	0.5
Lower internalisation OECD	700'000	280'000	2'317'000	0.3
Upper internalisation OECD	300'000	120'000	993'000	0.7
Central Non-OECD	800'000	320'000	2'648'000	0.2
Lower internalisation Non-OECD	1'000'000	400'000	3'310'000	0.0
Upper internalisation Non-OECD	500'000	200'000	1'655'000	0.5

7.1.3 Premature Morbidity

Much of the discussion that applies to valuation of mortality risks from accidents applies to the valuation of injuries. Unfortunately there is not a single study on which we can rely to provide us with baseline unit values. Therefore, we rely on the work of Lindberg (1999), who usefully summarizes the ratios between fatality values and values for severe[10] and minor[11] injuries. He concludes that the recommendation made by ECMT (1998), of weighting the risk value for severe injuries at 13% and for minor injuries at 1% of the risk value of fatalities is broadly supported by the evidence from individual, generally CVM, studies - though the studies reflect a wide range of values. These ratios – and the unit values they generate - are also consistent with injury values adopted in previous ExternE work. The unit values of injuries are reported in Table 12.

[10] Severe injuries include amputation, major fractures, serious eye injuries, loss of consciousness and any injuries requiring hospital treatment over 24 hours.

[11] Minor injuries include other accidents responsible for the loss of more than three working days.

However, whilst Lindberg (1999) splits injuries into “severe” and “minor” categories, the historical data on incidence of injuries resulting from fuel-cycle accidents does not disaggregate in this way. Consequently, the bottom line in the table presents unit values for a “typical” injury, represented by the mean of the “severe” and “minor” categories.

Table 12: Morbidity unit values (€ 2002).

	Central	Minimum	Maximum
Value of a Statistical Life	1’000’000	400’000	3’310’000
Severe injury	130’000	52’000	430’300
Minor injury	10’000	4000	33’100
"Typical injury"	70’000	28’000	231’700

7.1.4 Mental trauma

It is recognised that the mental trauma of being impacted by fuel-cycle related accidents might be a significant welfare effect in some instances. For example, should there be a hydro-electric dam breach in a given area, it is likely to affect those who live close by directly, by requiring them to move, in lieu of flood damage or indirectly because of their proximity and perceived vulnerability. Another example may be the trauma that follows from an oil platform accident that injures or kills other colleagues. There are therefore public and occupational valuation issues that need to be considered in this context.

The principal difficulty, with deriving monetary values for this impact category is that it is intangible and has psychological effects that cannot easily be identified or quantified in any meaningful way. It is therefore difficult to rank severity of mental trauma experiences and differentiate in monetary terms. This difficulty is combined with the fact that mental trauma is often experienced concurrently with a physical effect e.g. of injury or evacuation. To some extent, it would appear possible, in the case of physical injury, that mental trauma is being picked up in the valuation of the disutility component. In the context of evacuation, or proximity to a severe accident this is not so. One methodological possibility for valuing mental trauma is to multiply our mortality range values by a fraction determined by disability weightings that accord with individual mental health conditions. For example, the Dutch Disability Weights project gives a weighting of 0.76 of a life year lost to the condition of severe depression (Stouthard et al., 1997). However, there is no information available on the length of time associated with the mental trauma end-point. As a rough guide we suggest using a value of one year as reported since this is regarded as typical for flood damage victims.

7.1.5 Evacuation / Resettlement

Severe non-nuclear fuel cycle accidents such as hydroelectric dam failure and gas/oil leaks/spills have led to the temporary or permanent displacement of people from their homes and/or places of work. This clearly has welfare impacts and these might include tangible costs including damage to property and other economic assets, transport, food and accommodation costs, medical and miscellaneous costs, and subsequent income losses. Some of these costs (e.g. property, medical and employment) may have been internalised to the extent that private insurance payments cover these events. Intangible costs relate to disutility and may include mental trauma of the type noted above.

A survey of the literature has provided estimates of evacuation costs from the US, but not from the non-nuclear fuel cycle. Two studies, one from the context of a simulated radioactive evacuation, the other from the hurricane evacuation context has estimated unit values. The first, (Radioactive Waste Management Associates, 2000) makes estimates of direct economic costs using two categories: fixed evacuation costs of €180 per family. The second, (Tyndall Smith, 2000) gives the following mean approximate total costs of evacuation per household: €25 for accommodation, €50 food, €25 travel, €3 entertainment, and €5 miscellaneous, summing up to €108. No medical costs are included in this latter study. On the basis of this evidence, we use the transfer value range of €108 to €180 for fixed direct economic costs with a mid-point of €144.

There will also be the loss of output resulting from absenteeism for work over the length of evacuation period. A survey study in the UK (CBI, 1998) has calculated the direct cost of absence, based on the salary costs of absent individuals, replacement costs (i.e. the employment of temporary staff or additional overtime), and lost service or production time. This amounts to €88/day absence. We note, however, that indirect costs of absenteeism (i.e. costs relating to lower customer satisfaction and poorer quality of products or services leading to a loss of future business) are not included. The UK survey estimates that these are €160/day absence, though this value was based on a small sample size. Including both elements produces a total of €248/day absence – we suggest that this should be the maximum value in a range from €88/day absence. A mid-point of €168 is a central estimate. There is no estimate available for the dis-utility of suffering evacuation though this might be thought to be very substantial. Clearly, there is overlap with the discussion of mental trauma - for which, as noted above, WTP values are elusive.

Resettlement costs associated with the construction of dams, though these are in relation to countries outside the EU. These costs are presented in Table 13. Comparison, however, is limited by inconsistency with regard to the cost elements included in estimates for individual dams. For this reason, robust unit values are difficult to recommend and we therefore do not make any recommendations for this impact end-point.

Table 13: Resettlement costs from construction of dams - €, 2002. Source: Bartolome et al. (2000).

	Construction	Resettled	Resettlement €	Cost per person €
James Bay, Canada	1995	18’000	594’940’000	**33’052**
Akosombo, Ghana	1965	80’000	50’000’000	625
Theun Hinboum, Laos	1998	25’000	2’600’000	104
Iron Gate 1, Romania	1971	24’000	69’300’000	2888
Pak Mun, Thailand	1994	4945	23’000’000	4651
Kariba, Zimbabwe	1959	57’000	601’000	11
Nam Ngum, Laos	1972	3474	58’500’000	16’839
Lesotho Highlands WP	2017	8400	43’000’000	5119
Magat, Philippines	1983	2150	8'214’285	3821
Kotomale, Sri Lanka	1985	13’000	4’251’249	327
Hunan Lingjintan, China	1996	4275	28'140’678	6583
Shuikou, China	1993	84’400	209’547’000	2483
Average Non-OECD				**3950**

As with the welfare impacts of evacuation, these cost estimates do not include estimates for disutility. These could, in theory, be estimated using either contingent valuation or hedonic price techniques. We are not aware of any such estimates being made for this impact. We suggest that the unit values for evacuation should be adjusted by PPP for non-OECD countries in policy analysis. In the absence of specific country contexts it seems most sensible to use the un-adjusted values given here.

7.1.6 Ban on consumption of food

We might expect a welfare impact to result from changes in food commodity prices and quantities as a result of a ban on food consumption following a contamination incident. Such ban on consumption could be expected as a result of oil spills both in land and/or in aquatic biomes. Empirical estimates from the non-nuclear accident context are not easy to come by – indeed it is unlikely that estimates, were they available, would be transferable since there is likely to be a high degree of context specificity. However, whilst not related to the non-nuclear fuel chain, the compensation to farmers on beef ban in UK, presented in below, provides an illustration of the producer surplus element of the associated welfare loss. This could be used as an indicator of the costs magnitude involved in bans on consumption of food.

Cost of beef ban in the UK: In the Spring of 2001 the spread of foot and mouth disease in the UK led to the statutory precautionary slaughter of any cow and sheep herds who either contained diseased animals or who – through their vicinity – might have been carrying the disease. In April 2001, the British government announced a scheme for compensating beef, dairy and sheep farmers affected by the foot and mouth disease. Farmers received full market value for slaughtered animals. In addition, compensation was paid for any feeding stuffs or any other materials destroyed or seized as being possibly contaminated, which could not be satisfactorily disinfected.

The compensation scheme, approved by the European Commission on 3rd April 2001, involved payments of € 30 per head of cattle and € 2.2 per sheep. This gave a total of € 180 million initially and a further € 35 million for the beef sector in Autumn 2001 – equivalent to just under five percent of the total UK sectoral output.

7.1.7 Land Contamination

Costs of restoring land to the condition it was in before a fuel cycle accident can be estimated from existing experience of clean - up of areas that have been contaminated by similar substances that are likely to contaminate from fuel-cycle accidents. Of course it should be remembered that cost estimates such as these based on actual expenditures made represent minimum estimates of WTP values. WTP values may however be derived from the economic values that accrue to the owners of the land once the land is restored and put to economic use, above what they would have been in its contaminated condition. We have not been able to make assessments of appropriate unit values because of the lack of available data. Future work would – in any case – be best undertaken in specific contexts since this impact category does not lend itself to generic transfer of values.

7.1.8 Economic Losses

Economic losses are likely to result from severe accidents in addition to those identified in the categories above if e.g. business operations are disrupted. In principle, economic losses can be estimated by changes in market supply and demand conditions - partial equilibrium welfare analysis. As an example of estimates of economic losses due to oil spills we note a study conducted by Cohen (1995), who employed a market model to evaluate the economic losses of the 1989 Exxon Valdez oil spill on Alaska's fishery. The methodology used involved a three phase ex-post forecasting approach to estimate economic losses from the oil spill. First, the author estimated provisional values of the accident's harvest volume impacts in each of the fisheries affected. Second, initial estimates were derived of the ex-vessel prices of regionally harvested fish and shellfish that would have prevailed in the absence of the oil spill. Finally, the (econometric) analysis constructed several alternative simulations to isolate the accident's social costs from a number of confounding biological and economic factors.

Determination of the social costs of the Exxon Valdez oil spill on Alaska's fisheries involved estimating the difference between the economic benefits that would have been derived in the absence of the oil spill with those derived in the presence of the accident. The social costs of the oil spill on Alaska’s fisheries during 1989, based on the provisional estimates of the accident’s harvest volume and ex-vessel price impacts, were US$108.1 millions. In 1990, the oil spill ‘s social costs on Alaska’s fisheries were estimated to have been US$47.0 millions. As with land contamination impacts, we do not recommend the transfer of unit values based on these, or other, estimates due to the highly context-specific nature of such incidents.

7.1.9 Clean-up/repair costs and willingness to pay (WTP) for recreational/ecosystem losses - oil spills costs

The welfare impacts of oil spills are likely to be determined by the scale of the spill, the ecological services that the impacted area supports and the scale and nature of "human" related services affected in the area. Estimation of these welfare impacts has had a certain level of attention in the wake of a number of high profile oil spills - primarily in the Atlantic and North Sea regions. In theory, welfare valuation should be estimated by calculating the different components of Total Economic Value: Direct and Indirect/Passive Use plus Non-Use values. Economic assessments have been undertaken; the results for two are summarised below.

1996 Sea Empress oil spill - Atlantic, off the South Wales coast, UK.

Approximately 72’000 tonnes of crude oil and 480 tonnes of heavy fuel oil were released into the sea, and 100km of coastline were affected. Commercial and recreational fishing was banned for 7 months and the tourism industry was affected. Large numbers of marine organisms were killed whilst several thousand sea birds were killed. The financial and economic costs are summarised in Table 14.

Table 14: Summary of total costs resulting from Sea Empress oil spill (£m). Source: Environment Agency (1998).

	Financial costs		**Economic costs**	
Category	*Lower Bound*	*Upper bound*	*Lower Bound*	*Upper bound*
Direct clean-up costs	49.1	58.1	49.1	58.1
Tourism	4	46	0	2.9
Recreation	-	-	1.0	2.8
Commercial fisheries	6.8	10	0.8	1.2
Recreational fisheries	0.1	0.1	0.8	2.7
Local industry	0	0	0	0
Conservation/non-use	-	-	22.5	35.4
Human health	-	-	1.2	3
Total	60.0	114.3	75.3	106.1

Note that the lower and upper bounds reflect the uncertainty as to how to best ascribe measures of costs to the oil spill. Note also that the economic costs are greater than the financial costs for the conservation of ecosystems and their non-use values, reflecting the fact that these costs - whilst having welfare effects - are not reflected in financial market prices.

<u>1989 Exxon Valdez Oil Spill, Gulf of Alaska.</u>

Approximately 39’000 metric tonnes of crude oil was released in Prince William Sound, before spreading to the Gulf of Alaska - 1300 miles of coastline were oiled. There were acute damages to seabirds (250’000 dead), bald eagles, marine mammals and inter-tidal communities. Longer-term impacts were borne by Pacific herring, pink salmon and the inter-tidal and sub-tidal environments. Assessments of the impacts varied between scientists a decade after the event. The most detailed estimates of welfare impacts that exist derive from the compensation payments made by Exxon as a result of combined civil and criminal settlements. These payments included the following:

Civil Settlements

- WTP damage assessment (including passive use values, aesthetic and non-use measured by CVM), litigation and clean-up: €213 million
- Research, monitoring and general restoration: €180 million
- Habitat protection: €395 million
- Long term restoration: €108 million
- Science management, Public information and administration €31 million

Criminal settlements

- Habitat protection and improvements: €100 million

Total economic damage equated to €1.027 billion.

In order to derive unit damage values for future damage risk assessment, we can derive damage cost per tonne of oil in the two examples. This produces values of €26’333 and €2368 per tonne of crude oil for Exxon Valdez and Sea Empress, respectively. The difference can be explained partly by the fact that different elements of TEV were given attention in the two cases, partly by the fact that the damage in the case of oil spills is clearly contingent upon location and weather conditions at the time that determine dispersal patterns, and, of course, partly by different preferences between populations. For these reasons the most sensible course of action in making recommendations of unit values is to suggest a range of unit values that could be used in risk assessment exercises that might inform policy. The lower value, derived from the Sea Empress incident is in fact supported by evidence from a number of oil spills in the Caspian Sea that have resulted in average damage costs of €2600 per tonne. We therefore take this modal average as a central value. As a consequence we suggest that the best indicative unit values to use are:

– Central - €2600/tonne

– Minimum - €2300/tonne

– Maximum - €24’000/tonne

These are clearly not robust values to be relied upon in all contexts and we would not make any differentiation between OECD and non-OECD countries. Nevertheless these values provide a useful range with which to work.

7.1.10 Conclusions

The sections above have summarised the main evidence relating to the estimation of unit values that might apply to the monetisation of externalities arising from non-nuclear fuel cycle accident impacts. We have provided unit values for mortality and morbidity impacts as well as evacuation and damage from oil spills and they are collected in Table 15. It is clear that the evidence to support estimation of unit values for many of the impact categories considered is either of poor quality, of wide variance or non-existent. As a result, unit values that are presented make up ranges of values. These ranges would have to be used in full in subsequent policy analysis for the results to have credibility.

Table 15: Summary of results. Unit values for fuel cycle accident end-points (in €2002), provided for various levels of internalisation (expressed in parentheses).

	Central	Minimum	Maximum
Value of a Statistical Life	1’000’000	400’000	3’310’000
Occupational fatalities			
Central OECD (80%)	200’000	80’000	662’000
Lower internalisation OECD (70%)	300’000	120’000	993’000
Upper internalisation OECD (100%)	0	0	0
Central Non-OECD (50%)	500’000	200’000	1’655’000
Lower internalisation Non-OECD (0%)	1’000’000	400’000	3’310’000
Upper internalisation Non-OECD (100%)	0	0	0
Occupational injuries			
Central OECD (80%)	14’000	5600	46’340
Lower internalisation OECD (70%)	21’000	8400	69’510
Upper internalisation OECD (100%)	0	0	0
Central Non-OECD (50%)	35’000	14’000	115’850
Lower internalisation Non-OECD (0%)	70’000	28’000	231’700
Upper internalisation Non-OECD (100%)	0	0	0
Public fatalities			
Central OECD (50%)	500’000	200’000	1’655’000
Lower internalisation OECD (30%)	700’000	280’000	2’317’000
Upper internalisation OECD (70%)	300’000	120’000	993’000
Central Non-OECD (20%)	800’000	320’000	2’648’000
Lower internalisation Non-OECD (0%)	1’000’000	400’000	3’310’000
Upper internalisation Non-OECD (50%)	500’000	200’000	1’655’000
Public injuries			
Central OECD (50%)	35’000	14’000	115’850
Lower internalisation OECD (30%)	49’000	19’600	162’190
Upper internalisation OECD (70%)	21’000	8400	69’510
Central Non-OECD (20%)	56’000	22’400	185’360
Lower internalisation Non-OECD (0%)	70’000	28’000	231’700
Upper internalisation Non-OECD (50%)	35’000	14’000	115’850
Evacuation			
Fixed costs per household	144	108	180
Daily costs per household	168	88	248
Oil spills - welfare costs per tonne of oil	2600	2300	24’000

7.2 Damage costs and external costs of severe accidents in different energy chains

Damage costs and external costs of severe accidents in different energy chains were calculated, based on the energy-chain specific damages and unit values provided in chapter 7.1. The estimated damage and external costs for OECD-countries are considered to be also representative for EU-15.

For external costs, different degrees of internalization for occupational and public fatalities in OECD and non-OECD countries were applied. Values for injured and evacuees were similarly treated. Fixed costs of evacuees per household were converted to costs per person because ENSAD only contains information on the number of evacuated persons. Conversion factors applied were 2.5 for OECD countries and 4.4 for non-OECD countries (United Nations Centre for Human Settlements (HABITAT), 2001; Keilman, 2003). Similar values have been reported in a number of other studies (Boongarts, 2001; European Environmental Agency (EEA), 2001; United Nations Population Fund (UNFPA), 2001).

Tables 16 to 19 provide full chain results for fatalities, injured, evacuees, and oil spill welfare costs expressed in €-Cent(2002). For more detailed data with decomposition of costs into plants and rest of the chain stages we refer to Appendix B; the full report provides further details on individual chain stages (Burgherr et al., to be published)[12]. Since the costs provided in Table 16 only cover <u>immediate</u> fatalities it is of interest to relate them to the accident damage costs based on PSA for the Swiss nuclear power plant Muehleberg, which are dominated by the costs of <u>latent</u> fatalities. The mean value has been assessed at 1.2E-3 US-cents/kWh_e, with 5-th and 95-th percentiles at 1.0E-4 and 3.8E-3 US-cents/kWh_e; these results include damage costs of non-health effects (Hirschberg et al., 1998).

Generally, average external costs for non-OECD countries were clearly higher than for OECD countries. For fatalities, non-OECD was between 15 and 55 times greater than OECD, depending if the Banqiao/Shimantan dam failure is included or not. The respective difference for injured and evacuees was substantially lower, i.e., costs for non-OECD about 2.7 and 3.4 times higher, respectively; this is at least partially due to lower completeness of injury data for non-OECD countries. Regarding costs of oil spills, it should be noted that these estimates are based on few examples only (see chapter 7.1), and thus do not reflect at all spill specific conditions (compare Appendix A).

Concerning smaller accidents, no analysis at the level of detail performed for severe accidents was possible because of the much less comprehensive database. In spite of the

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[12] Such detailed decomposition of external costs is partially questionable in view of the scarcity of the corresponding statistical evidence.

substantial uncertainties involved smaller accidents appear to be minor contributors to the overall external costs of electricity generation. Gross estimates indicate that their share amounts to less than 10% of severe accident costs.

Table 16: Summary of full chain damage costs and external costs (€-Cents(2002)/kWh) of severe accidents with at least 5 fatalities. NA = not available. Value of a Statistical Life (central value) = 1.045 million Euro. Reference coal, oil and natural gas plants have efficiencies of 40%, 31% and 53%, respectively.

		Damage costs in €-Cents(2002)/kWh			External costs in €-Cents(2002)/kWh		
		Occupational	*Public*	*Total*	*Occupational*	*Public*	*Total*
Coal	*OECD*	1.70E-3	1.21E-5	1.71E-3	3.40E-4	6.06E-6	3.46E-4
	non-OECD w/o China	6.48E-3	4.32E-5	6.53E-3	3.24E-3	3.46E-5	3.28E-3
	China (1994-1999)	1.22E-2	NA	1.22E-2	6.10E-3	NA	6.10E-3
Oil	*OECD*	9.94E-4	9.02E-4	1.90E-3	1.99E-4	4.51E-4	6.50E-4
	non-OECD	1.82E-3	1.08E-2	1.26E-2	9.11E-4	8.66E-3	9.57E-3
Natural gas	*OECD*	2.24E-4	4.35E-4	6.59E-4	4.47E-5	2.18E-4	2.62E-4
	non-OECD	3.27E-4	5.89E-4	9.15E-4	1.63E-4	4.71E-4	6.34E-4
Hydro	*OECD*	NA	4.06E-5	4.06E-5	NA	2.03E-5	2.03E-5
	non-OECD	NA	1.23E-1	1.23E-1	NA	9.82E-2	9.82E-2
	non-OECD w/o Banqiao/Shimantan	NA	1.61E-2	1.61E-2	NA	1.29E-2	1.29E-2
Nuclear	*OECD*	NA	NA	NA	NA	NA	NA
	non-OECD	5.74E-4	NA	5.74E-4	2.87E-4	NA	2.87E-4

Table 17: Summary of full chain damage costs and external costs (€-Cents(2002)/kWh) of severe accidents with at least 10 injured. NA = not available. Value of a typical injury (central value) = 70'000 Euro. Reference coal, oil and natural gas plants have efficiencies of 40%, 31% and 53%, respectively.

		Damage costs in €-Cents(2002)/kWh			External costs in €-Cents(2002)/kWh		
		Occupational	*Public*	*Total*	*Occupational*	*Public*	*Total*
Coal	*OECD*	2.23E-5	NA	2.23E-5	4.45E-6	NA	4.45E-6
	non-OECD w/o China	5.31E-5	NA	5.31E-5	2.66E-5	NA	2.66E-5
	China (1994-1999)	3.37E-5	NA	3.37E-5	1.69E-5	NA	1.69E-5
Oil	*OECD*	1.00E-4	1.96E-4	2.96E-4	2.01E-5	9.79E-5	1.18E-4
	non-OECD	8.29E-5	5.36E-4	6.19E-4	4.14E-5	4.29E-4	4.70E-4
Natural gas	*OECD*	4.13E-5	1.08E-4	1.49E-4	8.27E-6	5.38E-5	6.21E-5
	non-OECD	1.77E-5	6.54E-5	8.31E-5	8.83E-6	5.23E-5	6.12E-5
Hydro	*OECD*	NA	1.56E-4	1.56E-4	NA	7.78E-5	7.78E-5
	non-OECD	NA	1.35E-5	1.35E-5	NA	1.08E-5	1.08E-5
	non-OECD w/o Banqiao/Shimantan	NA	1.35E-5	1.35E-5	NA	1.08E-5	1.08E-5
Nuclear	*OECD*	1.98E-5	NA	1.98E-5	3.96E-6	NA	3.96E-6
	non-OECD	4.59E-4	NA	4.59E-4	2.29E-4	NA	2.29E-4

Table 18: Summary of full chain damage costs and external costs (€-Cents(2002)/kWh) of severe accidents with at least 200 evacuees. NA = not available. Fixed evacuation costs per household (central value) = 144 Euro. Reference coal, oil and natural gas plants have efficiencies of 40%, 31% and 53%, respectively.

		Damage costs in €-Cents(2002)/kWh			External costs in €-Cents(2002)/kWh		
		Occupational	*Public*	*Total*	*Occupational*	*Public*	*Total*
Coal	*OECD*	NA	NA	NA	NA	NA	NA
	non-OECD w/o China	NA	NA	NA	NA	NA	NA
	China (1994-1999)	NA	NA	NA	NA	NA	NA
Oil	*OECD*	3.42E-7	8.26E-6	8.60E-6	6.84E-8	4.13E-6	4.20E-6
	non-OECD	NA	6.67E-6	6.67E-6	NA	5.34E-6	5.34E-6
Natural gas	*OECD*	2.19E-7	4.54E-6	4.75E-6	4.39E-8	2.27E-6	2.31E-6
	non-OECD	NA	9.46E-8	9.46E-8	NA	7.57E-8	7.57E-8
Hydro	*OECD*	NA	5.60E-6	5.60E-6	NA	2.80E-6	2.80E-6
	non-OECD	NA	2.11E-5	2.11E-5	NA	1.68E-5	1.68E-5
	non-OECD w/o Banqiao/Shimantan	NA	2.11E-5	2.11E-5	NA	1.68E-5	1.68E-5
Nuclear	*OECD*	NA	3.16E-5	3.16E-5	NA	1.58E-5	1.58E-5
	non-OECD	NA	7.83E-5	7.83E-5	NA	6.26E-5	6.26E-5

Table 19: Summary of oil spill welfare costs in €-Cents(2002)/kWh of severe accidents with at least 10’000 tonnes of hydrocarbons spilled. NA = not available. Oil spill welfare costs per tonne of oil: central value = 2600 Euro, minimum = 2300 Euro, maximum = 24’000 Euro.

	Damage costs in €-Cents(2002)/kWh		
	Central estimate	*Minimum estimate*	*Maximum estimate*
OECD	3.70E-3	3.27E-3	3.41E-2
non-OECD	5.50E-3	4.87E-3	5.08E-2

8. Conclusions

As a result of recent efforts building futher on earlier extensive investigations of energy-related accidents, the basis for the technical comparison of severe accident risks associated with different energy chains has been significantly improved. The advancements include in particular extension of the period of observation up to the year 2000, improved completeness of historical records, upgrades in quality and consistency of the information, and better coverage of various types of damages. The present work also generated new unit values for fuel cycle accident end-points, which in combination with the energy chain specific accident indicators, made it possible to estimate the corresponding external costs. These estimates are first-of-its-kind for the non-nuclear fuel cycles.

For the sake of completeness the following conclusions are provided for the major energy chains individually and what regards comparisons among the chains. Due to space limitations the basis for the energy chain specific conclusions has been only partially elaborated in the present report. For the full account, including compilations of the relevant datapoints we refer to Burgherr et al. (to be published).

Accident risks associated with the various stages of full energy chains were explicitly considered, unless it was clear that major risks are concentrated to one specific stage in the chain.

8.1 Specific energy chains

Coal chain

1. The overall number of severe (≥5 fatalities) accidents in the coal chain decreased in OECD countries in the last two decades as opposed to non-OECD countries. Additionally, very large accidents with more than 100 fatalities occurred less often in OECD countries than non-OECD counries in the 1980s and 1990s.
2. The number of fatalities in OECD countries decreased significantly. While the coal production was increased there has been a simultaneous reduction of severe accidents due to legislation, research findings concerning the prevention of gas and coal-dust explosions, fires and inundations, as well as closure of old unsafe mines.
3. The experience with accidents in the Chinese coal chain points to large differences compared to other non-OECD countries and thus needs to be analyzed separately.
4. More than every third industrial severe accident in China occurs in the coal industry. Every year about 6000 fatalities occur in Chinese mines due to small and severe accidents. Though severe accidents receive more attention than the small ones about 2/3 of the fatalities is due to the small ones.

5. The Chinese severe accident fatality rate for the coal chain exceeds 6 fatalities per $GW_e yr$. On average, this is about ten times higher than in non-OECD countries and about forty times higher than in OECD countries.
6. The coal chain stage with by far most fatalities is “Extraction”. The other stages are small contributors to severe accidents. In the industrialised world some smog catastrophes (e.g., Great London Smog in December 1952), which have features of severe accidents occurred in the 50s and 60s and have not been repeated since.
7. The most frequent cause for world-wide severe (≥5 fatalities) coal accidents are methane gas explosions in underground mining. Fires, roof collapses and transport accidents had significantly lower contributions.

Oil chain

1. OECD and non-OECD countries clearly showed opposite trends in number of accidents in the period 1969-2000. While the former decreased by almost 50%, the latter nearly doubled. In contrast, there is also a slight increase in number of fatalities for OECD countries, but at distinctly lower levels than for non-OECD countries.
2. The most risk prone stages in the oil chain are “Regional Distribution” and “Transport to Refinery”. About two thirds in OECD countries and close to three quarters in non-OECD countries of all severe (≥5 fatalities) accidents in the oil chain occurred in these two stages. Furtermore, the most severe accidents fatalities also occurred in these stages. In contrast, the more than 40 refinery accidents resulted in less than 40 fatalities per accident, except for one accident with 150 fatalities (Nigeria, 2000), when thieves were pumping gasoline from a vandalised pipeline at a refinery.
3. Maritime accidents are the most frequent accidents during the stage “Transport to Refinery” while road accidents are the most frequent accidents during the stage “Regional Distribution”. In the latter mentioned stage petrol is the primary oil product involved.
4. “Natural oil pollution” - such as seepage from the ocean bottom and oil releases from eroding sedimentary rocks - accounts for almost 50% of oil inputs to the sea. However, these large amounts are released at very low rates, so that surrounding ecosystems have adapted and even evolved to utilize some of the hydrocarbons. In terms of the quantities released, oil spills as a consequence of shipping, platform and pipeline accidents are less significant than oil spills caused by operational discharges (e.g., cargo washing), spills of non-tanker vessels, costal facility spills, and land-based sources (river and runoff).

5. 136 offshore and 39 onshore oil spills with hydrocarbon releases of at least 10’000 tonnes occurred between 1969 and 2000. Although the largest tanker spill only ranks on the fifth position, tanker accidents have accounted for most of the world’s largest oil spills.
6. The following “hot spots” for tanker oil spills have been identified (Etkin, 1997): Gulf of Mexico, northeastern US, Mediterranean Sea and Persian Gulf. Regarding offshore activities, the North-Sea is the most unfriendly environment, and consequently has a high share of severe offshore accidents (Hirschberg et al., 1998).
7. However, factors other than the quantity released (distance from the coast, weather and current conditions, time profile of the discharges and sensitivity of the areas exposed to oil pollution), contribute to and are often decisive in the context of the ecological disasters caused by tanker and platform accidents. For example, the Exxon Valdez spill is widely considered the number one spill worldwide in terms of damage to the environment, although it ranks only 45th among the largest tanker accidents.

Gas chain

1. The yearly number of LPG and natural gas severe (≥5 fatalities) accidents substantially increased after 1970 for non-OECD countries, whereas it remained at similar levels or even decreased in OECD countries. For fatalities, similar trends were observed, but at the same time there is a large scatter from year to year due to few very large accidents.
2. The majority of severe (≥5 fatalities) accidents occurred in transportation stages followed by “Heating” for natural gas, and “Regional Distribution” for LPG.
3. Nearly 57% of all severe (≥5 fatalities) natural gas accidents occurred during transport by pipelines, distantly followed by activities such as process (10.4%), storage (8.8%) and incidents that originated in domestic or commercial premises (Dom/com; 17.6%). The majority of accidents involving pipelines were caused by impact failures (46%) and mechanical failures (30%).
4. Almost half of all severe (≥5 fatalities) LPG accidents occurred during transport, particularly by road tankers. The dominant accident cause was impact failure.

Nuclear chain[13]

1. In the historical experience of nuclear reactor accidents two events are clearly dominant, namely the TMI-2 and Chernobyl accidents. While the first mentioned accident had practically negligible health and environmental consequences, the latter resulted in disastrous impacts. Preliminary estimates of these impacts have been reproduced in the present work. Having in mind their partially latent nature the definite assessment cannot be made at this stage.
2. Due to the radical differences in the plant design and operational environment the Chernobyl accident is essentially irrelevant for the evaluation of the safety level of the representative western nuclear power plants. This also applies to a large extent to most nuclear power plants in non-OECD countries.
3. Use of a plant-specific PSA, if available, is the most rationale basis for the estimate of consequences of severe accidents and the associated external costs. The results obtained from such an approach are by definition representative for the case being studied. In addition, it enables treatment of uncertainties in a transparent and disciplined way. In case this approach is not feasible, any extrapolation of results obtained for a specific plant in a specific environment must be done with great care; the reference case should be carefully selected with view to similarities in the design philosophy and in the operating environment. Some earlier published applications do not exhibit such a care.
4. Estimates of external costs of severe nuclear accidents show the largest discrepancies in the past studies and are considered controversial. Independently of the numerical results, use of the Chernobyl accident as the only reference for the assessment of environmental consequences is more than questionable. Generally, state-of-the-art, rationale and defensible methodological approaches, based on full scope PSAs, have not been used extensively in this context.
5. The results obtained for western plants using predominantly PSA-based approaches show low (quantifiable) contributions of severe accidents to external costs of nuclear power. This contrasts with some estimates based on simplistic, limited in scope and arbitrary approaches published in the literature. Low (absolute) contributions are to be expected as a reflection of the defence in depth design philosophy. In the particular case of the Swiss Muehleberg plant, the early offsite risks are negligible due to relatively low radionuclide inventory and low population density in the immediate

[13] Since the reference results for the nuclear chain originate from Hirschberg et al. (1998) the conclusions remain unchanged. No specific nuclear incidents during the last few years support essential modifications of these conclusions.

proximity of the plant. The extensive backfitting has been generally efficient in terms of reduction of the applicable risk measures. Generalisations should, however, be avoided - the indication is applicable to plants with good safety standards and within the limited boundaries of the analyses performed. The relative differences between the various applications can still be large since the risks are expected to be strongly plant- and site-specific.

Hydro chain

1. Depending on the evaluation time period and the related boundary conditions the variation between the failure rates (mean values) obtained for the different dam types corresponds to a factor of 6 to 23.
2. With only few exceptions, the dam failure rates have decreased significantly in time. This is due to a combined effect of technological developments (including replacement of masonry by concrete as the primary construction material around 1930 and on) and the impact of regulatory requirements. In most cases there is a significant decrease in failure rates when the first five years of operation after filling the dam are excluded from the evaluation. This observation is important since a majority of current dams have long operating history, far beyond five years.
3. The Swiss dams exhibit a number of favourable safety-related features. Of particular importance is the typically relatively low capacity of earth dams, which is a positive factor for the mitigation of accidents and for the limitation of the extent of potential damages. The failure rates (mean values) based on generic and probably conservative estimates are in the range of 10^{-5} to 10^{-4} per dam-year and show a variation by a factor of at most 4.3 between the various dam types. The lowest estimate was obtained for gravity dams. For gravity, arch, buttress and rockfill dams the mean values are close to the estimated upper bounds, while lower bands are up to two orders of magnitude lower. The available statistical material is most comprehensive for earth dams.
4. Dam failure rates are not only subject to variation with respect to the type of dam but depend also to some extent on the purpose of the dam. This may partially reflect the different safety standards within the various areas of dam applications but is also a result of the differences in the distributions of dam types within these diverse applications. In this context flood control and hydro power dams appear on average to be the best performers. The water supply dams have the highest average failure rates.
5. Dam consequence analyses cases considered in this work show strong dependence of the results obtained for dams situated in areas with substantial population at risk on the consequence models used and on the assumed warning times. Theoretical consequence models tend to result in significantly higher consequence estimates than experience-based models. Given reasonable warning times consequences of dam breaks can be

strongly reduced through evacuation of large parts of population at risk. This emphasizes the importance of monitoring/inspections and efficient alarm systems.

6. Similar to the nuclear case also the results of dam risk assessment are strongly case-specific, which calls for the implementation of predictive approaches. The present work proposes use of a simplified, resource-saving probabilistic approach. It avoids the very detailed modelling of accident propagation prior to dam break. In addition, it recognises the difficulties and inherent limitations in the estimation of the associated probabilities. Such an approach would partially build on further refinement of historical evidence, extensive use of structured expert judgement for delineation and rough estimation of accident frequencies associated with specific initiating events as well as for the estimation of the timing characteristics of such sequences, and on detailed consequence analysis. As a second element the proposed approach would in any case include the development of moderately detailed event trees; the expert judgement would be extensively used for the assessment of the branch probabilities. The realism of this evaluation would be examined in view of the perspective provided by the treatment utilising the available historical experience.

8.2 Comparative aspects on damages and external costs of severe accidents

1. The present work demonstrates that comprehensive historical experience of energy-related severe accidents is available and can be used as the basis for quantifying the corresponding damages and external costs.
2. The evaluation of the historical experience with energy-related accidents shows quite large numerical differences between the aggregated risk indicators obtained for the various energy chains, as well as between the corresponding frequency-consequence curves. Hydro power in non-OECD countries and upstream stages within fossil energy chains are most accident-prone; natural gas chain exhibits the lowest risks among the fossil chains.
3. Energy-related accident risks in non-OECD countries are distinctly higher than in OECD countries. Regional differences have been shown to be of utmost importance particularly for the nuclear and hydro chains. The expectation values for fatality rates due to severe accidents are lowest for western hydro and nuclear power. This is also reflected in correspondingly low external costs associated with severe accidents estimated using state-of-the-art methods. At the same time the extent of consequences of hypothetical extreme accidents is largest in the case of hydro and nuclear. Valuation of this aspect depends on stakeholder preferences, can be addressed in multi-criteria analysis and along with the issue of wastes affects in particular the ranking of nuclear power in the sustainability context (Hirschberg et al., 2000).

4. PSA perspective on severe accident risks is particularly important for energy chains whose risks are dominated by power plants, the historical experience of accidents is scarce or its applicability is highly restricted. These conditions are valid for most western hydro and nuclear power plants.

5. The focus of the current work has been on severe accidents. Nevertheless, it has been demonstrated on the basis of few selected cases that the cumulative damages caused by much less spectacular small accidents may for fossil energy chains be of the same order or even larger than those due to the severe ones. The available databases do not adequately cover small accidents. Extension of the corresponding knowledge basis would require quite large resources since a bottom-up approach would be necessary. It is not expected that the implementation of such an approach would result in significant increases of external costs in the absolute sense since at least in industrialised countries there is already a rather high level of internalisation of costs of small accidents.

6. Damages caused by severe accidents in the energy sector are rightly subject of concern but remain quite small compared to those caused by natural disasters. More important, though the estimates of external costs of energy-related accidents are still based on incomplete information for some of the end-points and are thus inherently non-conservative, the corresponding external costs are numerically rather insignificant when compared to the external costs of air pollution. This conclusion is reassuring what concerns the robustness of the overall external cost estimates

8.3 Recommendations on future developments

Having in mind the results but also limitations of the present work some recommendations on desirable future developments can be made. These recommendations are not made exclusively with view to improvements of accident-related external cost estimates but consideration is given to the more broader role the accident issue plays in the evaluation of current and future options for energy supply.

- It is in the nature of the topic that new accidents occur thus extending the historical experience. The corresponding databases need to be maintained, further extended and used for the estimation of updated risk indicators.

- The current analysis addressed the currently operating systems. Of interest is to investigate more extensively trends and use them to address the issue of the potential influence of technical advancements and improved operational safety on the risk performance of future systems.

- Improvements of specific indicators are desirable. Of particular importance is improved consistency and completeness of data on direct economic damages.

- For hydropower a demonstration application of simplified Probabilistic Safety Assessment (PSA) for few representative dam types and sites should be considered. This calls for close cooperation with dam experts.
- For the nuclear chain the available full scope results should be extended and updated. Application of a simplified PSA-approach to establish risk indicators for selected advanced design(s) at few representative sites in Europe is recommended.
- Small accidents have not been addressed in detail. Broader and systematic evaluation of such accidents particularly in the fossil chains is needed. This would require a rather large effort as the bulk of the relevant raw data is strongly decentralised.
- Valuation of some end-points and the degree of internalisation was based on quite limited literature sources. Extensions of the basis are probably feasible.
- It is unlikely that the issue of low probability-high consequence accidents will be resolved in the public arena by the fact that the corresponding estimates of external costs tend to be low. Risk aversion issues based on the estimated indicators need to be systematically addressed in integrated sustainability evaluations.
- Adding a ‘geo-referenced component’ to ENSAD, i.e. coupling with Geographic Information System (Arcgis), will be considered.

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GLOSSARY

BHDF
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CCIY
China Coal Industry Yearbook

CISDOC
International Occupational Health and Safety Centre Bibliographic Database

CRED
See EM-DAT

CVM
Contingent Valuation Method

DNV
Det Norske Veritas. See WOAD

EM-DAT
Since 1988 the WHO Collaborating Centre for Research on the Epidemiology of Disasters (CRED) has been maintaining an Emergency Events Database - EM-DAT. EM-DAT was created with the initial support of the WHO and the Belgian Government.

ENSAD
Energy-related Severe Accidents Database; this comprehensive database on severe accidents with emphasis on those associated with the energy sector has been established by the Paul Scherrer Institute, Switzerland.

ETC
The Environmental Technology Centre maintains a worldwide tanker spill database where accidental spills of over 1000 barrels of petroleum products were released. Incidents can be searched for by date and/or vessel name.

EU
European Union

ExternE
The ExternE project was the first comprehensive attempt to use a consistent 'bottom-up' methodology to evaluate the external costs associated with a range of different fuel cycles. The European Commission launched the project in collaboration with the US Department of Energy in 1991.

Final consumption
The term final consumption implies that energy used by the energy producing industries and for transformation is excluded

HSE
Health and Safety Executive (UK).

HSELINE
Library and Information Services of HSE

IAEA
Interanational Atomic Energy Agency

ICOLD
International Commission on Large Dams

IEA
International Energy Agency

ILO
International Labour Organisation

ITOPF
International Tanker Owners Pollution Federation Ltd.

LLP
Lloyd's Casualty Week; formerly Lloyd's of London Press

LPG
Liquefied Petroleum Gas

MARS
The Major Accident Reporting System is a distributed information network of the European Union

MHIDAS
Major Hazards Incidence Data Service

MSHA
Mine Safety and Health Administration (USA)

NIOSHTIC
National Institute of Occupational Safety and Health (USA)

OECD
Organisation for economic cooperation and development

OFDA
See EM-DAT

OSH
Occupational Health and Safety

PC-FACTS
Failure and Accidents Technical Information System; TNO Department of Industrial Safety, The Netherlands.

PSA
Probabilistic Safety Assessment

PPP
Purchase Power Parity

SIGMA
Sigma is published approximately eight times a year by Swiss Re's Economic Research & Consulting Team based in Zurich, New York and Hong Kong

TEV
Total Economic Value

TMI
Three Mile Island

VLYL
Value of Life year Lost

VSL
Value of Statistical Life

WOAD
Worldwide Offshore Accident Databank; Det Norske Veritas, Norway

WTA
Willingness to Accept

WTP
Willingness to Pay

UNITS

t	tonne, metric ton (1 t = 1000 kg)
Mt	one million tonnes or one megatonne (1 Mt = 10^6 t)
toe	tonnes of oil equivalent
tce	tonnes of coal equivalent
W	watt (1 W = 1 J/sec)
kW	kilowatt (1 kW = 10^3 W)
MW	megawatt (1 MW = 10^6 W)
GW	gigawatt (1 GW = 10^9 W)
kWh	kilowatt hour (1 kWh = 3.6 MJ
$GW_e yr$	gigawatt-year (1 $GW_e yr$ = 8.76 x 10^9 kWh)
J	joule (1 J = 1 Nm^{-1} = 1 $kgm^{-1}s^{-2}$)
MJ	megajoule (1 MJ = 10^6 J)
Bq	1 Becquerel = amount of material which will produce 1 nuclear decay per second. The Bequerel is the more recent SI unit for radioactive source activity. The curie (Ci) is the old standard unit for measuring the activity of a given radioactive sample. It is equivalent to the activity of 1 gram of radium. 1 curie = 3.7 x 1010 Becquerels.
Gy	Gray; SI unit of absorbed radiation dose in terms of the energy actually deposited in the tissue. The Gray is defined as 1 joule of deposited energy per kilogram of tissue. The old SI unit is the rad. 1 Gy = 1 J/kg = 100 rad.
Ryr	Reactor*year

APPENDIX A: OIL SPILLS

A.1 Background

Causes of oil spills include carelessness, natural disasters such as earthquakes or weather extremes as well as intentional events (terrorists, war, vandalism and dumping). Every day about 119 billion liters of oil are being transported at sea (Cutter, 2001). But not all spills come from tankers. Some originate from storage tanks, pipelines, oil wells, tankers and vessels cleaning out tanks.

A.2 Input of oil to the sea

Recently, the Committee on Oil in the Sea (National Research Council (NRC), 2003) has published updated estimates for average annual releases of petroleum inputs by source to the sea (Figure 21). *Natural seeps* are purely natural phenomena that occur when curde oil seeps from the geologic strate beneath the seafloor to the overlying water column. These seeps are the highest contributors of petroleum hydrocarbons to the marine environment. However, these large amounts are released at very low rates, so that surrounding ecosystems have adapted and even evolved to utilize some of the hydrocarbons (Spies et al., 1980; Spies & DesMarais, 1983; Montagna et al., 1986; Montagna et al., 1989). In other words, ecological impacts of seeps appear to be limited in area, but as a contaminant "background" it is important to determine the extent of pollution resulting from human activities. Extraction, transportation and consumption of petroleum include all significant sources of anthropogenic petroleum pollution.

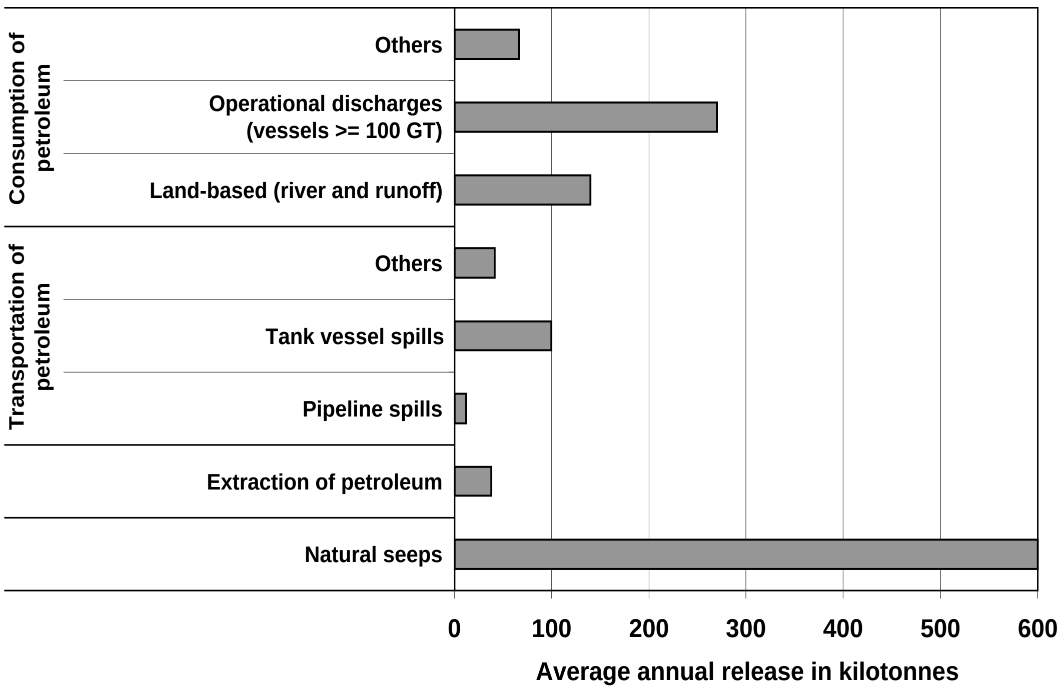

Figure 21: Average annual contributions (1990-1999) from major sources of petroleum in kilotonnes to worldwide marine waters.

The nature and size of releases due to *petroleum extraction* is highly variable, but is restricted to areas where active oil and gas exploration and development are under way. In the period 1985 to 2000, the number of offshore oil and gas platforms rose from a few thousand to about 8300 fixed or floating offshore platforms, following the increase in world oil production (National Research Council (NRC), 2003). Historically, the second largest marine spill worldwide was a blowout at the Ixtoc-I well that released 480’000 tonnes of crude oil into the Gulf of Mexico over a ten-month period (June 1979 – February 1980). However, improved production technologies and safety training of personnel have dramatically reduced accidental spills from platforms to about 3% of petroleum inputs worldwide.

Petroleum transportation can result in releases of dramatically varying sizes, from major spills associated with tanker accidents to relatively small operational releases that occur regularly. Although, releases from the transport of petroleum now amount to less than 13% worldwide, they can still have disastrous effects because ecological impacts are not simply depending on the quantity of hydrocarbons spilled, but is a complex function of distance to the coast, weather and current conditions among other factors. Finally, it should be noted that regional inputs to the sea may significantly differ from global estimates. For example, van Bernem & Lübbe (1997) report estimates of annual oil inputs for different regions: North Sea 260’000 t, Baltic Sea 21’000-66’000 t, Mediterranean Sea 500’000-1’000’000 t, Carribean Sea 950’000 t, Persian/Arabian Gulf 190’000[14], Arabian Sea ca. 5’000’000 t.

Petroleum consumption can result in releases as variable as the activities that consume petroleum. Yet, these typically small but frequent and widespread releases contribute the overwhelming majority of the petroleum that enters the sea due to human activity.

A.3 Oil Spill Trends

In total, 175 severe oil spills with at least 10’000 tonnes were recorded in the years 1969-2000 (Figure 22). However, it is apparent from the Figure that the majority of spills resulted in hydrocarbon releases of less than 5000 tonnes. Spills below 100 tonnes were not included because data on numbers and amounts are highly incomplete, but analyses by ITOPF (2003c) suggest that the vast majority of spills are very small (i.e., 85% of 10’000 accidents fall into the smallest category <7 tonnes).

[14] Recent numbers may be substantially higher due to the Iran-Iraq war and the 1991 Gulf War.

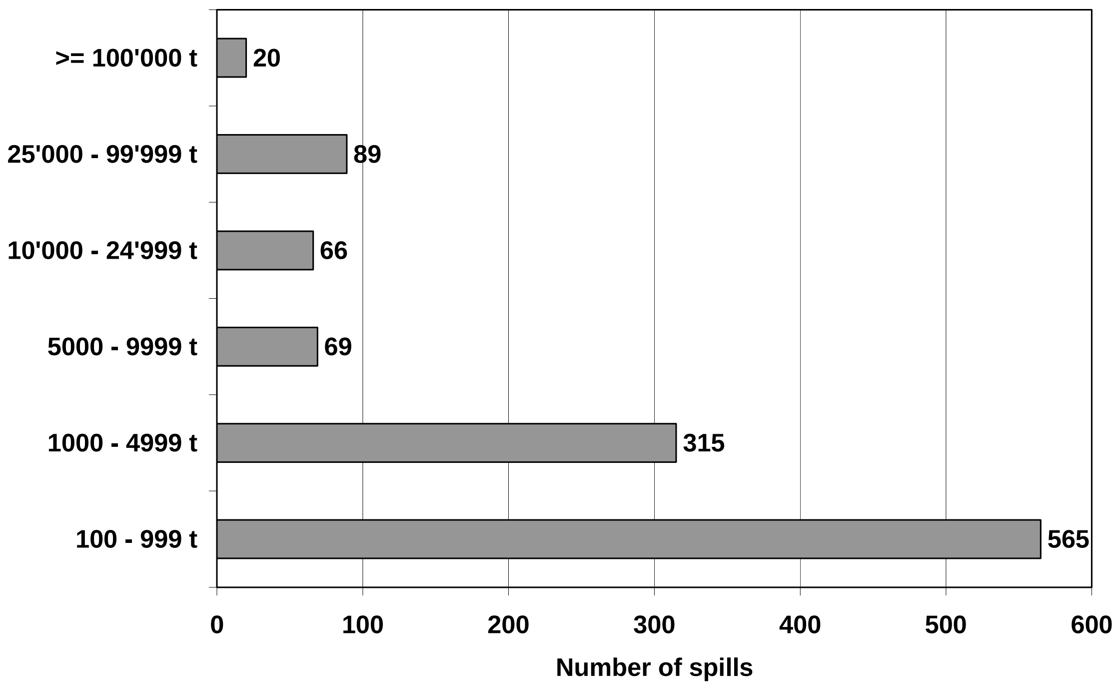

Figure 22: Distribution of the number of oil spills for the period 1969-2000.

Figure 23 shows the number of severe (≥10’000 tonnes) offshore and onshore oil spills for the period 1969-2000. Overall, 136 offshore and 39 onshore spills were recorded. Offshore oil spills showed an increasing trend between 1969-1979, followed by a decrease of more than 50% for the decade averages for the 1980s and 1990s. In contrast, onshore spills remained at similar levels over the whole period of observation.

It is notable that a very few extremely large spills are responsible for a high percentage of the oil spilt (Figure 24). For example, 6 spills over 100’000 tonnes out of a total 40 spills accounted for 50% of the oil spilt in the ten-year period of 1980-1989. The figures for a particular year may therefore be severly distorted by a single large accident. This is cleary illustrated by 1978 (Amoco Cadiz), 1979 (Atlantic Empress / Aegan Captain, Ixtoc-1 Platform), 1983 (Castillo de Bellver, Nowruz 4 Platform) or 1991 (ABT Summer).

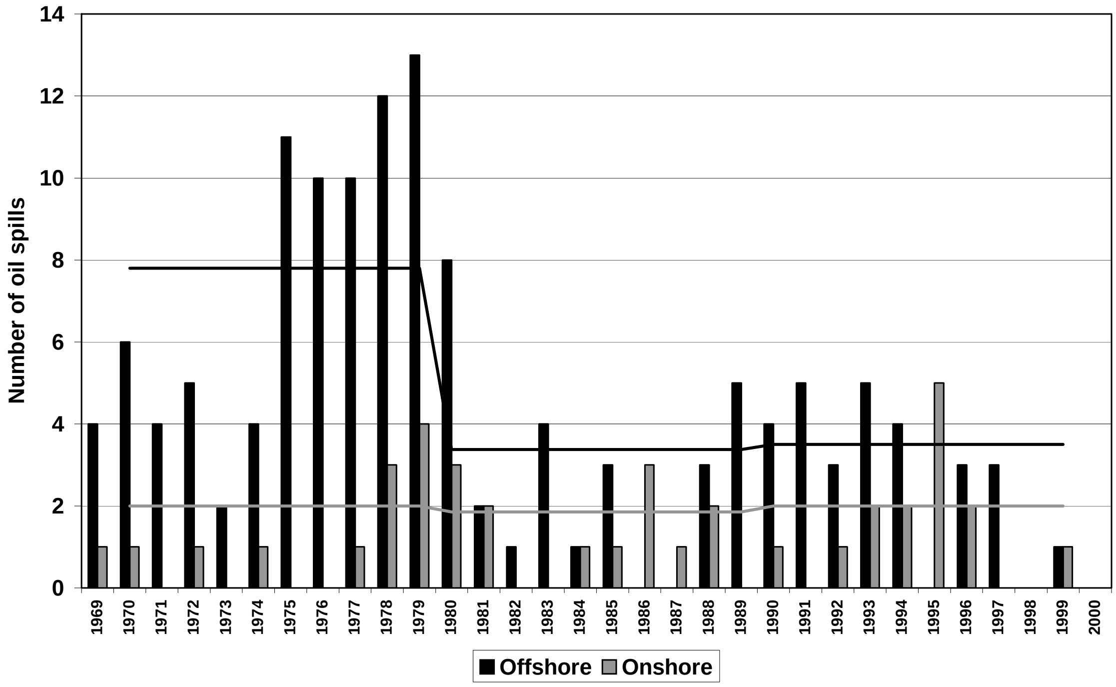

Figure 23: Number of severe (≥10'000 tonnes) offshore and onshore oil spills for the period 1969-2000.

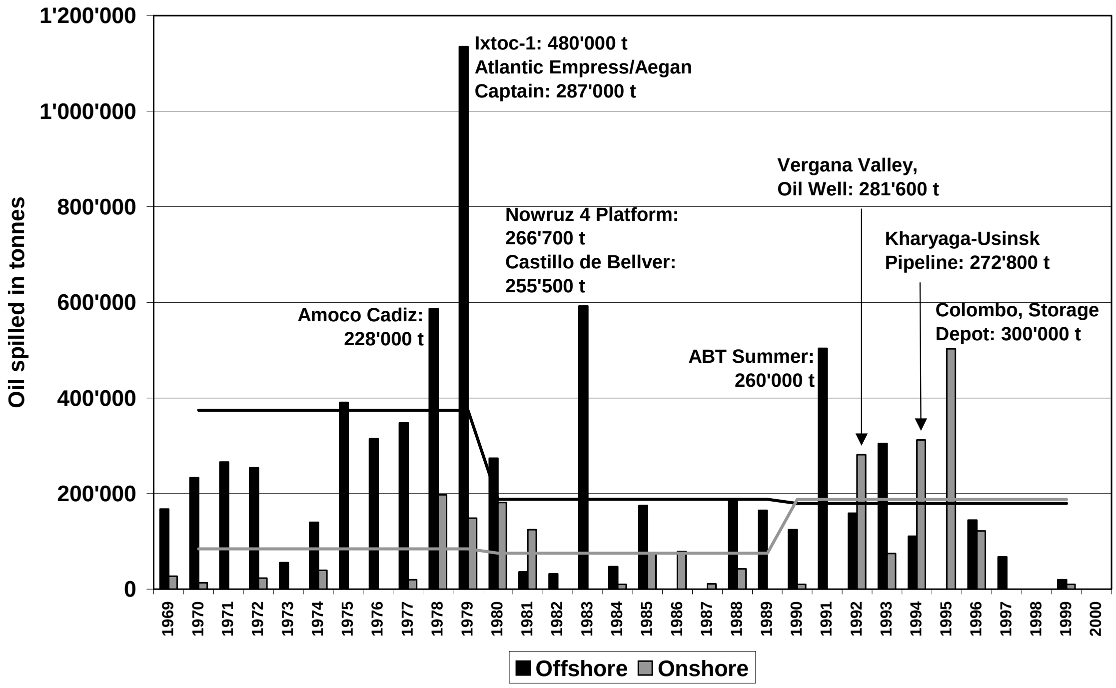

Figure 24: Amounts of oil spilt in offshore and onshore accidents for the period 1969-2000. Note that the Gulf War II spill in 1991 is not shown.

Table 20 summarizes the top ten oil spills that occurred in the period 1969-2000. The biggest spill ever occurred during Gulf War II in 1991 when between 768’000 and 1’770’000 tonnes spilled from oil terminals and tankers. The second biggest spill happened over a ten-month period (June 1979 - February 1980) when 480’000 tonnes spilled at the Ixtoc I well blowout in the Gulf of Mexico near Ciudad del Carmen (Mexico). In comparison, the largest tanker spill had a size of about 290’000 tonnes (Sea Empress / Aegean Captain; 1979).

Table 20: The top ten oil spills that occurred in the period 1969-2000.

Year	Country	Location	Description	Oil spilled (tonnes)
1991	Kuwait	Mina al-Ahmadi and Sea Island Terminal	Over a period of about 4 months crude oil was released into the Arabian Gulf as part of Gulf War II	1’770’000
1979	Mexico	Gulf of Mexico, Bahia de Campeche	Blow-out of deep exploratory well IXTOC-1	480’000
1994	Russian Federation	Usinsk, Kolva River tributary	Spill of Kharyaga-Usinsk Pipeline	300’000
1995	Sri Lanka	Colombo	Storage tanks at two depots destroyed by bomb attacks	300’000
1979	Trinidad and Tobago	off Tobago	Spill of supertankers Atlantic Empress (Greece reg.) and Aegean Captain (Liberia reg.) after collision	287’000
1992	Uzbekistan	Fergana Valley	Blow-out of oil well	281’600
1983	Islamic Republic of Iran	Nowruz oil field	Blow-up of offshore oil field during Gulf War I	266’700
1991	Angola	off coast Angola	Explosion and fire on tanker ABT Summer (Liberia reg.)	260’000
1983	South Africa	Atlantic, off Saldanha Bay, Cape Town	Fire on the tanker Castillo de Belliver (Spain reg.)	255’500
1978	France	Brittany, off Portsall	Spill of tanker Amoco Cadiz (Liberia reg.)	228’000

Figure 25 shows that there is considerable variation by spill source, for both the number and size of severe (≥10’000 tonnes) oil spills for the period 1969-2000. Tanker spills dominate the picture with shares of about 74% for the number of spills and about 64% for spill sizes, respectively. However, the percentage of oil contributed by tanker spills has decreased from 74% in the 1970s to 52% in the 1990s. In contrast, shares from Refinery/Storage Tank and Pipeline sources have substantially increased, accounting for 17.2% and 20.4% of spill amounts in the 1990s.

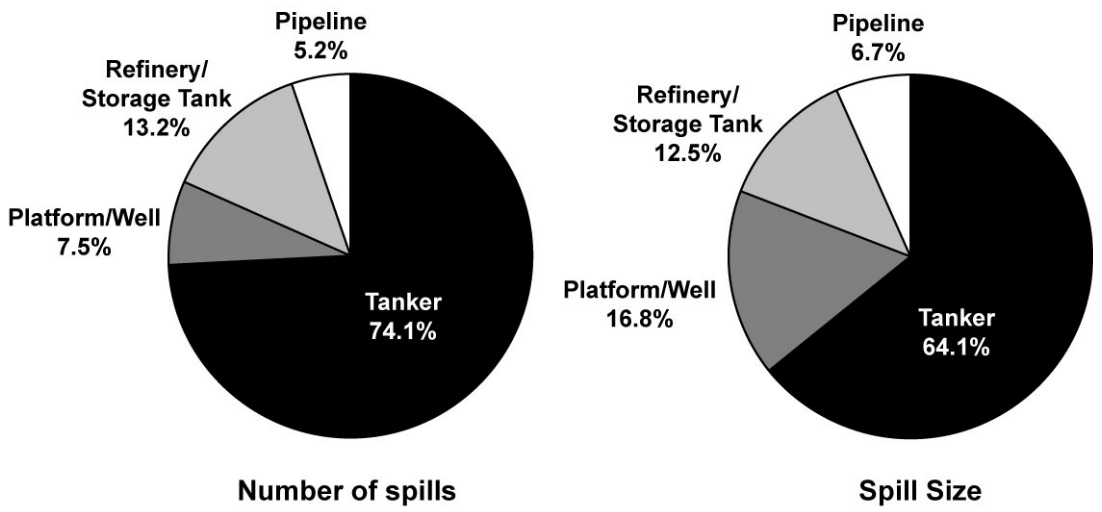

Figure 25: Percentages of oil spill numbers and size according to various sources for the period 1969-2000.

Contrary to increases in oil movement and to popular perceptions after recent catastrophic events, the number of spills and total spillage of tanker accidents have decreased significantly since the 1970s (Figure 26). This decrease may be for several reasons.The enactment of the Oil Polluion Act of 1990 placed increased liability on responsible parties, and other regulations required the phase out of older vessels and the implementation of new technology and safety procedures (National Research Council (NRC), 2003). While the statistics show encouraging downward trends, there is no room for complacency: (1) spills that occur in sensitive locations still cause devastating ecological and economic impacts, and (2) cleanup costs have risen dramatically in the last two decades.

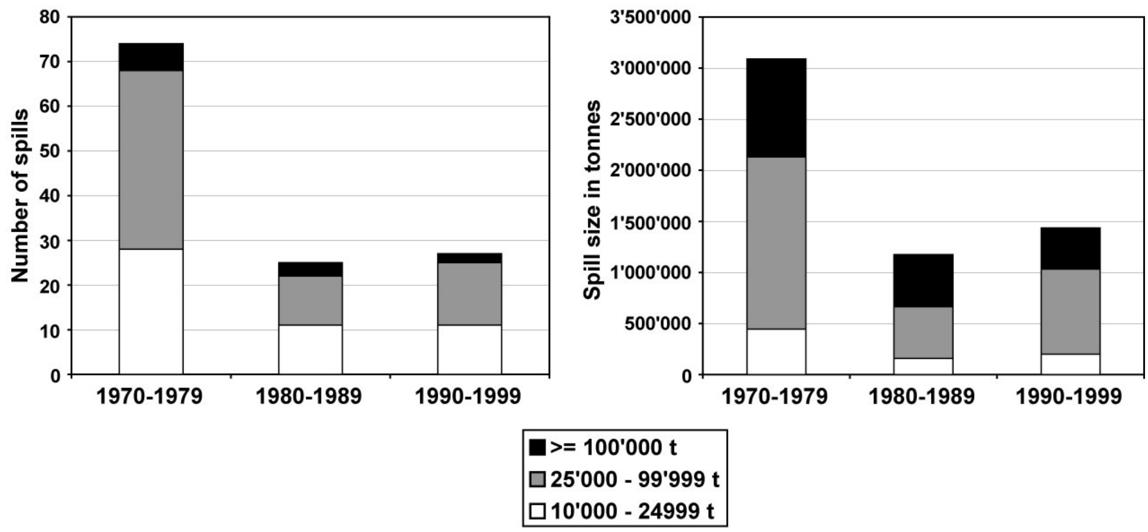

Figure 26: Number of oil spills and spill size in tanker accidents for the period 1969-2000.

A.4 Ecological impacts and socio-economic factors affecting the cost of oil spills

The following discussion is focused on oil spills from tanker accidents because these events often result in potentially high impacts and costs, and thus receive high-profile attention by the public, media, politicians, regulators and claimants. However, it should be noted that oil spills from other sources can also have large impacts. For example, the blow-out of the Ixtoc-1 well offshore Mexico in 1979 resulted in a total damage of 411 million USD (Sharples, 1992).

The ecological and socio-economic impacts and the resulting cost of tanker spills vary considerably from one accident to another, depending on a number of interrelated factors. These factors include:

- Type of oil
- Amount spilled and rate of spillage
- Spill location

Additionally, the effectiveness of the clean-up is also influenced by the quality of the contingency plan as well as the management and control of actual response operations.

<u>Type of oil</u>

Heavy fuel and crude oils are generally of low toxicity, but they are highly persistent, which mainly results in physical contamination. Furthermore, these oils have the potential to travel great distances from the original spill location. As a consequence, the clean-up can be extremely difficult, include large areas and be costly. For example, the Nakhodka (Japan, 1997) and Erika (France, 1999) spilled relatively small amounts of 19’000 t and 20’000 t of fuel oil, but its persistency resulted in maximum spreading and widespread coastal contamination (White, 2002; White & Molloy, 2003). As a consequence, compensation was settled at approximately 219 million USD for the Nakhodka, whereas claims are still being processed for the Erika, but are likely to considerabely exceed 180 million USD (ITOPF, 2003a).

In contrast, light refined products (e.g., gasoline, diesel) tend to be more toxic, but do not persist on the surface of the sea for a long time due to evaporation and easy dispersion and dissipation. In the case of the Braer incident (Shetland Isles (UK), 1993) the entire cargo of 85’000 t was dispersed by the rough weather conditions so that shoreline contamination was minimal (White, 2002; White & Molloy, 2003). In relative terms, costs were also relatively low with 83 million USD (ITOPF, 2003a).

In general, there is evidence that responses to spills of heavy fuels are more than 10 times more expensive than for lighter crudes and diesel fuels (Etkin, 2000).

Amount spilled and rate of spillage

The amount of oil spilled is clearly an important factor in determining impacts and costs. Nevertheless, the three largest tanker spills of the Atlantic Empress/Aegean Captain off Trinidad & Tobago in 1979 (287’000 t), ABT Summer off Angola in 1991 (260’000 t) and Castillo de Bellver off South Africa in 1983 (255’500 t) resulted in relatively low clean-up and damage costs because coastlines were not contaminated (White, 2002; White & Molloy, 2003). Several studies suggest that cleanup cost per tonne is significantly negatively correlated with spill size because of the costs associated with setting up a cleanup operation (Monnier, 1994; Etkin, 2000).

The rate of spillage is also a major factor. Continous releases over a longer time period from a damaged tanker close to the coast may require repeated clean-up efforts and could lead to long-term effects on fishery resources or tourism.

Spill location

The location of a spill can have considerable effects because it determines the severity of damage to the environment and economic resources as well as the requirement and extent of the clean-up. Regarding proximity to the shore, Etkin (2000) showed that nearshore spills and in-port spills are 4-5 times more expensive to clean up than offshore spills. However, spill location is not simple a surrogate for distance to the coast, it also includes local conditions such as weather conditions, water currents and depths, and tidal range. The vulnerability of different shoreline types is another site-specific factor (van Bernem & Lübbe, 1997). Ecosystems also exhibit differences in persistence and resilience following disturbance, resulting in different recovery trajectories. Finally, sensitivities are affected by seasonal differences in prevailing organisms and community structure at the specific time of a pollution event.

Consequently, the various factors associated with location are often of primary importance for impacts to the marine environment (Hirschberg et al., 1998; National Research Council (NRC), 2003). For example, the Exxon Valdez accident (Prince William Sound, Alaska, USA) was relatively small with 37’000 t oil lost, but it occurred close to the coastline and wind current moved the oil slick to the beaches leading to an ecological disaster. For instance, the resource damage figures indicate that between 100’000 and 300’000 birds (mostly guillemots, *Uria* sp.), 1500 to 5000 sea otters (*Enhydra lutris*), 300 harbor seals (*Phoca vitulina*), 250 bald eagles (*Haliaeetus leucocephalus*), up to 22 killer whales (*Orcinus orca*), and billions of salmon and herring eggs perished (van Bernem & Lübbe, 1997; Exxon Valdez Oil Spill Trustee Council, 2003). Cleanup costs alone amounted to about 2.5 billion USD, and total costs (including fines, penalties and claims settlements) are estimated at 9.5 billion USD (ITOPF, 2003a).

Besides effects on marine life, oil spills can (1) contaminate fishing equipment and mariculture facilities, (2) lead to temporary bans that affect commercial fishing, (3) cause loss of market confidence in marine products, and (4) and in some cases the depletion of fish stocks; particularly when spawning grounds are affected during spawning season, as it was the case in the Exxon Valdez spill.

Finally, oil spills can interfere with the normal operation of power stations and desalination plants that require a continous supply of clean seawater, and with the safe operation of coastal industries and ports (ITOPF, 2003b).

In conclusion, there is no simple answer to the question “How much does it cost to clean up an oil spill?”. However, a sound understanding of the complex array of interacting factors is crucial that contingency planners, response officials, government agencies and oil transporters can develop high-quality spill prevention programs and realistic oil spill contingency plans that are also cost-efficient. For a monetary evaluation of oil spills expressed as damage costs per tonne and a transfer of these unit values to different welfare values see chapter 7.

APPENDIX B: SELECTED EXTERNAL COST RESULTS FOR SEVERE ACCIDENTS

Table B1: External costs of severe accidents with at least 5 fatalities for the different energy chains are given for the period 1969-2000. Values are reported for power plant stage, rest of chain and total chains, respectively. Final consumption of fossil chains was available in Mtoe, therefore the following efficiencies were used for reference plants: 0.40 for coal, 0.31 for oil and 0.53 for gas. For hydro and nuclear this conversion step could be omitted because final consumption was already given in GWh_e. The central value of a Statistical Life applied in this project is 1'045'000 million Euro(2002). Values were not adjusted for Purchase Power Parity (PPP), except for China, for which a correction factor of x0.21 was used (compare chapter 7). The following degrees of risk internalization were used: 0.8 for occupational and 0.5 for public fatalities in OECD countries, and 0.5 and 0.2 in non-OECD countries, respectively. Estimates for external costs are then finally given in €-Cents/kWh_e. ng = neglible.

	Power plant		Rest of chain		Total chain
Energy Chain	Origin	€-Cents/kWhe	Origin	€-Cents/kWhe	€-Cents/kWhe
Coal	OECD	1.06E-6	OECD	3.45E-4	3.46E-4
	OECD	1.06E-6	non-OECD w/o China	3.28E-3	3.28E-3
	non-OECD w/o China	ng	non-OECD w/o China	3.28E-3	3.28E-3
	China	ng	China	6.10E-3	6.10E-3
Oil	OECD	6.01E-7	OECD	6.49E-4	6.50E-4
	OECD	6.01E-7	non-OECD	9.47E-3	9.47E-3
	non-OECD	9.81E-5	non-OECD	9.47E-3	9.57E-3
Natural gas	OECD	1.48E-6	OECD	2.61E-4	2.62E-4
	OECD	1.48E-6	non-OECD	6.34E-4	6.36E-4
	non-OECD	ng	non-OECD	6.34E-4	6.34E-4
Hydro	OECD	2.03E-5	OECD	ng	2.03E-5
	non-OECD	9.82E-2	non-OECD	ng	9.82E-2
	non-OECD w/o Banqiao/Shimantan	1.29E-2	non-OECD w/o Banqiao/Shimantan	ng	1.29E-2
Nuclear	OECD	ng	OECD	ng	ng
	non-OECD	2.87E-4	non-OECD	ng	2.87E-4

Table B2: External costs of severe accidents with at least 10 injured for the different energy chains are given for the period 1969-2000. Values are reported for power plant stage, rest of chain and total chains, respectively. Final consumption of fossil chains was available in Mtoe, therefore the following efficiencies were used for reference plants: 0.40 for coal, 0.31 for oil and 0.53 for gas. For hydro and nuclear this conversion step could be omitted because final consumption was already given in GWh_e. The central value of a "Typical Injury" applied in this project is 70'000 Euro(2002). Values were not adjusted for Purchase Power Parity (PPP), except for China, for which a correction factor of x0.21 was used (compare chapter 7). The following degrees of risk internalization were used: 0.8 for occupational and 0.5 for public fatalities in OECD countries, and 0.5 and 0.2 in non-OECD countries, respectively. Estimates for external costs are then finally given in €-Cents/kWh_e. ng = neglible.

	Power plant		Rest of chain		Total chain
Energy Chain	Origin	€-Cents/kWhe	Origin	€-Cents/kWhe	€-Cents/kWhe
Coal	OECD	3.86E-7	OECD	4.07E-6	4.45E-6
	OECD	3.86E-7	non-OECD w/o China	2.66E-5	2.69E-5
	non-OECD w/o China	ng	non-OECD w/o China	2.66E-5	2.66E-5
	China	ng	China	1.69E-5	1.69E-5
Oil	OECD	ng	OECD	1.18E-4	1.18E-4
	OECD	ng	non-OECD	4.29E-4	4.29E-4
	non-OECD	4.11E-5	non-OECD	4.29E-4	4.70E-4
Natural gas	OECD	2.13E-6	OECD	6.00E-5	6.21E-5
	OECD	2.13E-6	non-OECD	6.12E-5	6.33E-5
	non-OECD	ng	non-OECD	6.12E-5	6.12E-5
Hydro	OECD	7.78E-5	OECD	ng	7.78E-5
	non-OECD	1.08E-5	non-OECD	ng	1.08E-5
	non-OECD w/o Banqiao/Shimantan	1.08E-5	non-OECD w/o Banqiao/Shimantan	ng	1.08E-5
Nuclear	OECD	3.96E-6	OECD	ng	3.96E-6
	non-OECD	2.29E-4	non-OECD	ng	2.29E-4

Table B3: External costs of severe accidents with at least 200 evacuees for the different energy chains are given for the period 1969-2000. Values are reported for power plant stage, rest of chain and total chains, respectively. Final consumption of fossil chains was available in Mtoe, therefore the following efficiencies were used for reference plants: 0.40 for coal, 0.31 for oil and 0.53 for gas. For hydro and nuclear this conversion step could be omitted because final consumption was already given in GWh_e. The central value for fixed evacuation costs per household applied in this project is 144 Euro(2002). Fixed costs of evacuees per household were converted to costs per persons because ENSAD only contains information on the number of evacuated persons. Conversion factors used were 2.5 for OECD countries and 4.4 for non-OECD countries (United Nations Centre for Human Settlements (HABITAT), 2001; Keilman, 2003). Values were not adjusted for Purchase Power Parity (PPP), except for China, for which a correction factor of x0.21 was used (compare chapter 7). The following degrees of risk internalization were used: 0.8 for occupational and 0.5 for public fatalities in OECD countries, and 0.5 and 0.2 in non-OECD countries, respectively. Estimates for external costs are then finally given in €-Cents/kWh_e. ng = neglible.

	Power plant		Rest of chain		Total chain
Energy Chain	Origin	€-Cents/kWhe	Origin	€-Cents/kWhe	€-Cents/kWhe
Coal	OECD	ng	OECD	ng	ng
	OECD	ng	non-OECD w/o China	ng	ng
	non-OECD w/o China	ng	non-OECD w/o China	ng	ng
	China	ng	China	ng	ng
Oil	OECD	ng	OECD	4.20E-6	4.20E-6
	OECD	ng	non-OECD	4.57E-6	4.57E-6
	non-OECD	7.68E-7	non-OECD	4.57E-6	5.34E-6
Natural gas	OECD	7.43E-9	OECD	2.30E-6	2.31E-6
	OECD	7.43E-9	non-OECD	7.57E-8	8.31E-8
	non-OECD	ng	non-OECD	7.57E-8	7.57E-8
Hydro	OECD	2.80E-6	OECD	ng	2.80E-6
	non-OECD	1.68E-5	non-OECD	ng	1.68E-5
	non-OECD w/o Banqiao/Shimantan	1.68E-5	non-OECD w/o Banqiao/Shimantan	ng	1.68E-5
Nuclear	OECD	1.58E-5	OECD	ng	1.58E-5
	non-OECD	6.26E-5	non-OECD	ng	6.26E-5

VII REVISION OF EXTERNAL COST ESTIMATES

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List of Abbreviations

AA	Asthma attacks
AM	Acute Mortality
AOT40	Accumulated exposure over a threshold of 40 ppb
CHF	Congestive heart failure
CM	Chronic Mortality
CO_2	Carbon dioxide
CO_{2equiv}	Carbon dioxide equivalent
CVA	Cerebrovascular hospital admissions
ERF	Exposure-response function
HH	Human health
MRAD	Minor restricted activity days
nd	No data available
ng	Negligible
NMVOC	Non-methane volatile organic compounds
NO_x	Nitrogen oxides
nq	Not quantified
nu	Not used
RAD	Restricted activity days
RHA	Respiratory hospital admissions
SO_2	Sulphur dioxide
WEC	Wind energy converter
VLYL	Value of life year lost
YOLL	Years of life lost
VOLY	Value of Life Year
$VOLY_{chronic}$	Value of Life Year – chronic effects
$VOLY_{acute}$	Value of Life Year – acute effects
$VOLY_{disc}$	Discounted Value of Life Year
$VOLY_{undisc}$	Undiscounted Value of Life Year

1 Introduction

The various ExternE projects have produced an extensive database on external cost estimates from electricity generation, which are actively used in research and policy consultancy work. It has been beyond the scope of this project to provide a general update of the extensive database on previous external cost estimates, but nevertheless an indication is given on how existing external cost estimates will be affected by using the new or extended methodology developed in this and former projects.

Based on an inventory of the respective priority emissions, in a first step those electricity generating technologies covered in ExternE projects have been identified for which the new methodological developments, carried out in this project, are of particular importance. The effect of the new/extended methodology on the total external cost estimates is then analysed by applying the new methodological elements developed in this project to a small set of key technologies that have been analysed before in ExternE. These technologies include coal and oil fired plant and combined cycle plant using natural gas in four countries of the EU.

Since technologies have developed more rapidly in the renewable energy sector than for fossil power plants, it does not make much sense at this point (as originally planned in this project) making new calculations for those photovoltaic plants and wind turbines that have been assessed in the National Implementation phase – the criticism might arise to have used unfavourable results of renewable energy systems that are now far from being today’s state of technology. In the ongoing project ExternE-POL, however, one focus is the life-cycle analysis of several new and future technologies especially including renewables.

It has to be emphasised that the project is dedicated to show the outcomes of the improved methodology. This means, that the same specifications of the power plants, e.g. emission data, as for the ‘National Implementation’ project in 1997 (except some additional updated data for power plants in France for comparison), have been used, although the emissions of the power plants of course would have been changed for several reasons. So, the differences in the results of the National Implementation in 1997 and the new calculations lead to some general conclusions on how the new methodology affects current external cost estimates.

There has been more than one step of improvements of the methodology between National Implementation and NewExt. After the National Implementation there was for example the ExternE CoreTransport (Friedrich and Bickel 2001), followed by the project GREENSENSE (European Commission 2003a).

In this detailed final report, a further distinction is made between the three statuses of Externe: the results which are based on the state-of-the-art of the methodology used for National Implementation, those before NewExt (i. e., GREENSENSE) and those due to NewExt.

2 Exposure-response functions used for the National Implementation, before NewExt and for NewExt

Applying *EcoSense* the impacts of following pollutants are assessed:

SO_2, NO_x, primarily emitted PM_{10}, NMVOCs, secondary particles including deposition of N, S and acids from SO_2 and NO_x emissions, and tropospheric O_3 from NO_x and NMVOC emissions. These pollutants have an impact on different receptors. The assessed receptors are crops, building material and human health. The exposure-response functions (ERF) used are described in the following. More detailed descriptions can be found in the methodology descriptions of the ExternE project series in (European Commission 1999b) and (Friedrich and Bickel 2001).

2.1 Impact Assessment for Crops

For National Implementation there was no assessment of yield losses of rice, sunflower seed and tobacco. Apart from that the same impacts as described below have been used.

2.1.1 Effects from SO_2

The function for effects from SO_2, recommended in ExternE (European Commission 1999b) is adapted from one derived by (Baker et al. 1986). The function assumes that yield will increase with SO_2 from 0 to 6.8 ppb, and decline thereafter. The function is used to quantify changes in crop yield for wheat, barley, potato, sugar beet, and oats, and is defined as

$$y = 0.74 \cdot [SO_2] - 0.55 \cdot [SO_2]^2 \qquad \text{for } 0 < [SO_2] < 13.6 \text{ ppb}$$
$$y = -0.69 \cdot [SO_2] + 9.35 \qquad \text{for } [SO_2] > 13.6 \text{ ppb}$$

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

2.1.2 Effects from Ozone

For the assessment of ozone impacts, a linear relation between yield loss and the AOT 40 value (Accumulated Ozone concentration above a Threshold of 40 ppbV) calculated for the growth period of crops (May to June) is assumed (Fuhrer 1996). The relative yield loss change is calculated using the following equation together with and the sensitivity factors given in Table 1:

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

with y = relative yield change
α = sensitivity factors

Table 1: Sensitivity factors for different crop species

Sensitivity	α	Crop species
Slightly sensitive	0.85	rye, oats, rice
Sensitive	1.7	wheat, barley, potato, sunflower seed
Very sensitive	3.4	tobacco

2.1.3 Acidification of Agricultural Soils

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

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

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

2.1.4 Fertilisational Effects from Nitrogen Deposition

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

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

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

2.2 Impact Assessment for Building Material

The exposure-response functions used for impact assessment and recommended for ExternE (Friedrich and Bickel 2001) are listed below for different building materials. Apart from the exposure-response functions for carbonate paint (Haynie 1986), all are based on results from the UN-ECE ICP Materials (Kucera et al. 1997).

In a two-step approach, the exposure-response functions link the ambient concentration or deposition of pollutants to the rate of material corrosion, and the rate of corrosion to the exposure time of the material. Performance requirements determine the point at which replacement or maintenance is considered to become necessary. This point is given in terms of critical degradation. By entering the critical degradation into the formula and solving the equation for the reciprocal exposure time, the maintenance frequency is calculated.

2.2.1 Limestone

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

maintenance frequency: $1/t = [\,(2.7[SO_2]^{0.48}e^{-0.018T} + 0.019Rain[H^+])/R_{crit}\,]^{1/0.96}$

with
- R surface recession in µm
- $1/t$ maintenance frequency in 1/a
- $[SO_2]$ SO_2 concentration in µg/m^3
- T temperature in ^oC
- Rain precipitation in mm/a
- $[H^+]$ hydrogen ion concentration in precipitation in mg/l
- R_{crit} critical surface recession, European average value of 4000 µm

2.2.2 Sandstone, Natural Stone, Mortar, Rendering

surface recession: $R = (2.0[SO_2]^{0.52}e^{f(T)} + 0.028Rain[H^+]) \cdot t^{0.91}$

maintenance frequency: $1/t = [\,(2.0[SO_2]^{0.52}e^{f(T)} + 0.028Rain[H^+])/R_{crit}\,]^{1/0.91}$

with
- R surface recession in µm
- $1/t$ maintenance frequency in 1/a
- $[SO_2]$ SO_2 concentration in µg/m^3
- T temperature in ^oC
- $f(T)$ $f(T) = 0$ if $T < 10\ ^oC$; $f(T) = -0.013(T-10)$ if $T > 10\ ^oC$
- t time in years
- Rain precipitation in mm/a
- $[H^+]$ hydrogen ion concentration in precipitation in mg/l
- R_{crit} critical surface recession, European average value of 4000 µm

2.2.3 Zinc and Galvanised Steel

mass loss: $ML = 1.4[SO_2]^{0.22}e^{0.018Rh}e^{f1(T)}t^{0.85} + 0.029Rain[H^+]t$

maintenance frequency: $1/t = 0.14[SO_2]^{0.26}e^{0.021Rh}e^{f2(T)}/R_{crit}^{1.18} + 0.0041Rain[H^+]/R_{crit}$

with
- ML mass loss in g/m^2
- $1/t$ maintenance frequency in 1/a
- $[SO_2]$ SO_2 concentration in µg/m^3
- Rh relative humidity in %
- T temperature in ^oC
- $f1(T)$ $f1(T) = 0.062(T-10)$ if $T < 10\ ^oC$; $f(T) = -0.021(T-10)$ if $T > 10\ ^oC$
- $f2(T)$ $f2(T) = 0.073(T-10)$ if $T < 10\ ^oC$; $f(T) = -0.025(T-10)$ if $T > 10\ ^oC$
- t time in years
- Rain precipitation in mm/a
- $[H^+]$ hydrogen ion concentration in precipitation in mg/l
- R_{crit} critical surface recession, country-specific values

2.2.4 Paint on Steel

degradation rating: $A = (0.033[SO_2] + 0.013Rh + f(T) + 0.0013Rain[H^+])t^{0.41}$
maintenance frequency: $1/t = [(0.033[SO_2] + 0.013Rh + f(T) + 0.0013Rain[H^+])/A_{crit}]^{1/0.41}$

with	A	degradation rating, originally A=(10-ASTM), with ASTM representing a rating between 1 and 10 (10 = unexposed)
	1/t	maintenance frequency in 1/a
	$[SO_2]$	SO_2 concentration in $\mu g/m^3$
	Rh	relative humidity in %
	T	temperature in ^oC
		f(T) $f(T) = 0.015(T-11)$ if $T < 11\ ^oC$; $f(T) = -0.15(T-11)$ if $T > 11\ ^oC$
	Rain	precipitation in mm/a
	$[H^+]$	hydrogen ion concentration in precipitation in mg/l
	A_{crit}	the rating at which maintenance should occur, European value: 5

2.2.5 Paint on Galvanised Steel

degradation rating: $A = (0.0084[SO_2] + 0.015Rh + f(T) + 0.00082Rain[H^+])t^{0.43}$
maintenance frequency: $1/t = [(0.0084[SO_2] + 0.015Rh + f(T) + 0.00082Rain[H^+])/A_{crit}]^{1/0.43}$

with	A	degradation rating, originally A=(10-ASTM), with ASTM representing a rating between 1 and 10 (10 = unexposed)
	1/t	maintenance frequency in 1/a
	$[SO_2]$	SO_2 concentration in $\mu g/m^3$
	Rh	relative humidity in %
	T	temperature in ^oC
	f(T)	$f(T) = 0.04(T-10)$ if $T < 10\ ^oC$; $f(T) = -0.064(T-10)$ if $T > 10\ ^oC$
	Rain	precipitation in mm/a
	$[H^+]$	hydrogen ion concentration in precipitation in mg/l
	A_{crit}	the rating at which maintenance should occur, European value: 5

2.2.6 Carbonate Paint

material loss: $\Delta R = 0.12\ (1 - \exp(-0.121Rh/(100\text{-}Rh)))[SO_2] + 0.0174Rain[H^+]$
maintenance frequency: $1/t = (0.12\ (1 - \exp(-0.121Rh/(100\text{-}Rh)))[SO_2] + 0.0174Rain[H^+])/R_{crit}$

with	R	annual surface recession in µm/a
	1/t	maintenance frequency in 1/a
	$[SO_2]$	SO_2 concentration in $\mu g/m^3$
	Rh	relative humidity in %
	Rain	precipitation in mm/a
	$[H^+]$	hydrogen ion concentration in precipitation in mg/l
	R_{crit}	critical surface recession, country specific values

2.3 Impact Assessment for Human Health

Most exposure-response functions used for the National Implementation were also used before and are still in use within NewExt. However, some important ERF used in the National Implementation have already been changed before NewExt. In particular these are the ERF for nitrates (regarding human health), chronic bronchitis and 'chronic YOLL'. Moreover, the ERF for 'cases of chronic bronchitis' regarding children are not used any more.

ERF for ozone were also available. However, for National implementation there was no model implemented into EcoSense for evaluation of the increment of ozone concentration due to emissions of NO_x and NMVOC. Hence, for impacts via ozone, an estimation of the damages of ozone has been carried out within the ExternE Core Project, and had provided an average for the whole of Europe of 1,500 ECU_{1995}/t of NO_x emitted. (European Commission 1999a). Now, marginal changes in ozone concentration is calculated.

ERF for nitrates

During the EU-project GreenSense it was suggested by (Searl 2002) to scale down the ERF for nitrates with regard to human health by a factor of 0.5.

ERF for chronic bronchitis and cough (asthmatic children and asthmatic adults)

In ExternE Core/Transport the ERF for chronic bronchitis was scaled down by a factor of 0.5. The reason for this was the transfer of epidemiological studies from the US to Europe.

'Chronic YOLL'

In National Implementation the percent change in annual mortality rate/(µg/m3) for 'chronic mortality' was used to develop the ERF for 'chronic YOLL'. This value, which was applied only to adults older than 30 years, amounts to 72 YOLL per 100,000 persons older than 30 years per 1 µg PM_{10}/m^3 . The share of people older than 30 years was 57 % of the total population. In a later stage (GARP II) the ERF was recalculated in order to apply it to the entire population, which results in a value of 47 YOLL per 100,000 persons per 1 µg PM_{10}/m^3. In ExternE Core/Transport this value was again rescaled. Firstly, because of a transfer of epidemiological studies from the US to Europe by a factor of 0.5. Secondly, because of a different history of the exposure by a factor of 0.67. Overall the value for 'chronic YOLL' was scaled down by a factor of 1/3. Hence, the value for PM_{10} was 15.7 YOLL per 100,000 persons per 1 µg PM_{10}/m^3 . For nitrates this value is multiplied by 0.5, and for sulphates by a factor of 1.67.

Apart from the progress in the NewExt project, new insights within the concerted action DIEM suggest now to a factor of 39 YOLL per 100,000 persons per 1 µg PM_{10}/m^3, whereas the scaling factor for nitrates remain the same. Sulphates are now treated in the same way as PM_{10} particles.

The important exposure-response functions which have changed during the phase of NewExt are shown in Table 2.

Table 2: Exposure-response functions which have changed

Receptor	Impact Category	Reference	Pollutant	f_{er}
total	Chronic Mortality (CM)	(Pope et al. 2002)	PM_{10} Nitrates Sulphates PM2.5	0.320% 0.160% 0.320% 0.800%
adults	Chronic bronchitis	(Abbey et al. 1995)	PM_{10} Nitrates Sulphates	4.9E-5 2.45E-5 4.9E-5
adults asthmatics	Cough	(Dusseldorp et al. 1995)	PM_{10}, Nitrates Sulphates	0.335 0.168 0.335
children asthmatics	Cough	(Pope and Dockery 1992)	PM_{10} Nitrates Sulphates	0.267 0.133 0.267

It has to be emphasized that the exposure-response functions for chronic mortality are still under discussion. For example, at present the World Health Organisation (WHO) would have the general coefficient for $PM_{2.5}$ applied to all the components of $PM_{2.5}$, regardless of primary or secondary particles.

The assessed effects on human health and the applied exposure-response functions (ERF) are displayed in Table 3. The individual impact categories are explained and described in detail in (European Commission 1999b). The used ERF were taken from (European Commission 1999b) with changes based on recommendations by (Searl 2002) and (Hurley 2004).

Table 3: Quantification of human health impacts due to air pollution1) – used for National Implementation (NI), before and for NewExt; nu = not used

1) The exposure response slope, f_{er}, has units of [cases/(yr-person-µg/m^3)] for morbidity, and [%change in annual mortality rate/(µg/m^3)] for mortality. Concentrations of SO_2, PM_{10}, sulphates and nitrates as annual mean concentration, concentration of ozone as seasonal 6-h average concentration.

Receptor	Impact Category	Reference	Pollutant	f_{er} NI	f_{er} before NewExt	f_{er} NewExt
ASTH-MATICS						
Adults	Bronchodilator usage	(Dusseldorp et al. 1995)	PM_{10}	0.163	0.163	0.163
			Nitrates	0.163	0.082	0.082
			Sulphates	0.272	0.272	0.163
	Cough	(Dusseldorp et al. 1995)	$PM_{10,}$	0.168	0.168	0.335
			Nitrates	0.168	0.084	0.168
			Sulphates	0.280	0.280	0.335
	Lower respiratory symptoms (wheeze)	(Dusseldorp et al. 1995)	PM_{10}	0.061	0.061	0.061
			Nitrates	0.061	0.031	0.031
			Sulphates	0.101	0.101	0.061
Children	Bronchodilator usage	(Roemer et al. 1993)	PM_{10}	0.078	0.078	0.078
			Nitrates	0.078	0.039	0.039
			Sulphates	0.129	0.129	0.078
	Cough	(Pope and Dockery 1992)	PM_{10}	0.133	0.133	0.267
			Nitrates	0.133	0.067	0.133
			Sulphates	0.223	0.223	0.267
	Lower respiratory symptoms (wheeze)	(Roemer et al. 1993)	PM_{10}	0.103	0.103	0.103
			Nitrates	0.103	0.052	0.052
			Sulphates	0.172	0.172	0.103
All	Asthma attacks (AA)	(Whittemore and Korn 1980)	O_3	4.29E-3	4.29E-3	4.29E-3
ELDERLY 65+	Congestive heart failure (CHF)	(Schwartz and Morris 1995)	PM_{10}	1.85E-5	1.85E-5	1.85E-5
			Nitrates	1.85E-5	9.25E-6	9.25E-6
			Sulphates	3.09E-5	3.09E-5	1.85E-5
CHILDREN	Chronic cough	(Dockery et al. 1989)	PM_{10}	2.07E-3	2.07E-3	2.07E-3
			Nitrates	2.07E-3	1.04E-3	1.04E-3
			Sulphates	3.46E-3	3.46E-3	2.07E-3

Receptor	Impact Category	Reference	Pollutant	f_{er} NI	f_{er} before NewExt	f_{er} NewExt
CHILDREN	Cases of Chronic bronchitis (in Vol. 7 and Vol. 9: Chronic Bronchitis)	Dockery	PM_{10} Nitrates Sulphates	1.61E-3 1.61E-3 2.69E-3	nu nu nu	nu nu nu
ADULTS	Restricted activity days (RAD)[a)]	(Ostro 1987)	PM_{10} Nitrates Sulphates	0.025 0.025 0.042	0.025 0.013 0.042	0.025 0.013 0.025
	Minor restricted activity days (MRAD)[b)]	(Ostro and Rothschild 1989)	O_3	9.76E-3	9.76E-3	9.76E-3
	Chronic bronchitis	(Abbey et al. 1995)	PM_{10} Nitrates Sulphates	4.9E-5 4.9E-5 7.8E-5	2.45E-5 1.23E-5 3.9E-5	4.9E-5 2.45E-5 4.9E-5
ENTIRE POPU-LATION						
	Chronic Mortality (CM)	(Pope et al. 1995) (Pope et al. 2002)	PM_{10} Nitrates Sulphates	0.39% 0.39% 0.64%	0.129% 0.065% 0.214%	0.320% 0.160% 0.320%
	Respiratory hospital admissions (RHA)	(Dab et al. 1996)	PM_{10} Nitrates Sulphates	2.07E-6 2.07E-6 3.46E-6	2.07E-6 1.04E-6 3.46E-6	2.07E-6 1.04E-6 2.07E-6
		(Ponce de Leon et al. 1996)	SO_2 O_3	2.04E-6 3.54E-6	2.04E-6 3.54E-6	2.04E-6 3.54E-6
	Cerebrovascular hospital admissions (CVA)	(Wordley et al. 1997)	PM_{10} Nitrates Sulphates	5.04E-6 5.04E-6 8.42E-6	5.04E-6 2.52E-6 8.42E-6	5.04E-6 2.52E-6 5.04E-6
	Symptom days	(Krupnick et al. 1990)	O_3	0.033	0.033	0.033
	Acute Mortality (AM)	(Anderson et al. 1996)/ (Touloumi et al. 1996)	SO_2	0.072%	0.072%	0.072%
		(Sunyer et al. 1996)	O_3	0.059%	0.059%	0.059%
ADULTS 30+	Chronic YOLL	(Pope et al. 1995)	PM_{10} Nitrates Sulphates	7.2E-4 7.2E-4 12.0E-4	nu nu nu	nu nu nu

a) 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)

b) 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 asthmatics.
Thus, net MRAD = MRAD – (AA * 0.8 * 0.035)

The net restricted activity days (netRAD) need to be evaluated, because the number of restricted activity days (RAD) include cerebrovascular hospital admissions (CVA), congestive heart failure (CHF) and respiratory hospital admissions (RHA). There would be a double counting of RAD if these diseases would also be counted as RAD.

The exposure-response functions that have changed before and during the phase of NewExt (according to Table 3) refer to the primary pollutant PM_{10} and the secondary pollutants nitrates and sulphates. These changes as a whole are summarized in the following Table 4. At large, there has been a decrease of exposure-response function factors for chronic mortality, chronic bronchitis, and other respiratory health impacts, dependent on the type of pollutant causing the effect, and increases only for cough of asthmatics caused by PM_{10} and sulphates.

Table 4: Changes of exposure-response functions (ERF), described as quotient of ERF factors with the NewExt methodology (2004) versus National Implementation (1999a)

Pollutant / Human health impact	**PM_{10}**	**Nitrates**	**Sulphates**
Chronic mortality	0.82	0.41	0.5
Chronic bronchitis	1	0.5	0.63
Cough of asthmatics	2	1	1.20
Other respiratory health impacts	1	0.6	0.5

For the National Implementation an exposure-response function for 'chronic YOLL' was used which has to be applied to adults older than 30 years (ADULTS 30+). This group has a share of 57% of the population. Other exposure-response functions with regard to adults correspond to 80% of the population.

The terms 'acute' and 'chronic' relate to the time over which exposure to air pollution is relevant. 'Acute' relates to short-term exposures, hence 'acute mortality' relates to deaths that are brought forward as a result of pollution exposure over a period of days. 'Chronic' relates to problems of long-term exposure. Most of the air-pollution epidemiology carried out so far has concentrated on acute effects as these are easier to observe. A study can be set up in a relatively short period and results gained from observing pollution levels at existing monitoring stations and various health impacts for perhaps a year. In contrast, analysis of chronic effects clearly demands access to long term data sets, relatively few of which are available. One of the most notable studies in this field, that by (Pope et al. 1995), used data from the American Cancer Survey, which followed a large number of individuals for many years. One consequence of the problems of carrying out studies on effects of long-term exposures is that the extent to which available exposure-response functions can be thought to fully describe the health effects of air pollution is not clear. In particular, it may be expected that there are chronic effects through ozone exposure that have yet to be identified.

The slope derived from (Pope et al. 2002) was used within life table calculations to derive the Years of Life Lost (YOLL) per increase of $1 \mu g/m^3$ pollutant concentration. This new results are used for NewExt. For unspecified primary particles (PM_{10}) and sulfates a factor of

39 YOLL per increase of 1µg/m^3 was assessed (Hurley 2004). As for all other human health effects, for Nitrates half of the factor of PM_{10} was taken.

The functions given in Table 3 are applied within EcoSense to different risk groups of the population. The shares of population representing the different groups are given Table 5.

Table 5: Fraction of population referred to in Table 3

Population group	Fraction of population
Above 65 years	0.14
Adults	0.80
Adults 30+	0.57
Asthma adults	0.028
Asthma children	0.007
Children	0.2
Asthmatics	0.035
Total	1

3 Monetary Valuation of Impacts used for National Implementation, before NewExt and for NewExt

Several methods for the valuation of impact have been carried out. The methods are listed in Table 6.

Table 6: Methods for valuation

Market goods	For non-market goods (public goods, human health risks):	
Market prices only for goods traded on markets! (e.g. crops, timber)	Indirect evaluation methods Hedonic pricing (wage differences due to risks, price changes of houses or rents due to difference in air pollution or noise), Travel costs, prevention costs	Direct evaluation methods Contingent valuation (CVM), contingent ranking

It is only in limited cases that the values of goods and services damaged can be taken from market prices, such as the loss of agricultural crops, or building repair costs. Moreover, even in ‘simple’ cases, for example relating to reduced crop yield by air pollutants, it is far from trivial which prices to take. The decision taken here is to base crop valuations on world market prices rather than regional ones as they will be less distorted by subsidies. This presents no difficulty for products where there is world-wide trading of considerable importance (wheat, barley), but does create difficulty in cases where trading is carried out on a non-global basis. So, the selection which suitable prices to take has to be decided specifically from case to case.

For many impacts, however, there are no real market prices because the effects to be valued refer to public goods (health and the natural environment) and represent ‘intangible’ costs (effects on human health etc.). In these situations, the basis of valuation is the twin concepts of individual willingness to pay (WTP) for a reduction of a pollutant or damage or the willingness to accept (WTA) an increase in pollution or damage. In the case of valuing mortality impacts, estimates of the willingness to pay (WTP) for a reduction in risk or the willingness to accept (WTA) an increase in risk have been made by three methods. First, there are studies that look at the increased compensation individuals need, other things being equal, to work in occupations where the risk of death at work is higher. This provides an estimate of the WTA. Second, there are studies based on the contingent valuation method (CVM), where individuals are questioned about their WTP for measures that reduce the risk of death from certain activities (e.g. driving); or their WTA measures that, conceivably, increase it (e.g. increased road traffic in a given area). Third, researchers have looked at actual voluntary expenditures on items that reduce the risk of death, such as purchasing air bags for cars. The two concepts, WTP and WTA, imply different assumptions about the distribution of property rights for the environmental goods, so they are in general not equivalent. Empirical results show that willingness-to-accept is generally higher than willingness-to-pay. In ExternE and other externality studies most emphasis is placed on WTP as it is the more ‘conservative’ approach and has less of an inherent risk of exaggeration, although in principle both approaches are of equal standard.

Further details of the methods used to quantify health and other impacts are presented in reports available from the European Commission (European Commission 1999b).

The main changes in NewExt are the use of the NewExt survey results. The new values, are shown in Table 7.

Table 7: Monetary values for acute and chronic mortality

Age group	Impact: Human health	Value (EURO 2000)
Total	Acute YOLL	75,000
Total	Chronic YOLL	50,000

In Table 8 the monetary values that were used for National Implementation, for the projects running before NewExt and for NewExt are displayed. They are separated into the impact categories human health, crop and building materials.

Table 8: Monetary values used for economic valuation for National Implementation (European Commission 1999b) and (European Commission 1999d), values used before NewExt (Friedrich and Bickel 2001) and values used for NewExt

Age group	Impact: Human health	Value ECU_{1995}	Value $€_{2000}$ before NewExt	Value $€_{2000}$ NewExt
Above 65 years	Congestive heart failure	7,870	3,260	3,260
Adults	Chronic bronchitis	105,000	169330	169330
Adults	Minor restricted activity days (MRAD)	45	45	45
Adults	Restricted activity days	75	110	110
Asthma adults	Bronchodilator usage	37	40	40
Asthma adults	Cough	7	45	45
Asthma adults	Lower respiratory symptoms	8	8	8
Asthma children	Bronchodilator usage	37	40	40
Asthma children	Cough	7	45	45
Asthma children	Lower respiratory symptoms	8	8	8

All asthmatics	Asthma attacks (AA)	75	75	75
Children	Chronic cough	225	240	240
Children	Cases of Chronic Bronchitis	225	nq	nq
Total	Cerebrovascular hospital admissions	7,870	16,730	16,730
Total	Respiratory hospital admissions	7,870	4,320	4,320
Total	Symptom days	45	45	45
Total	Acute YOLL (3%)	155,000	165,700	75,000
Total	Chronic YOLL (3%)	84,330	96,500	50,000

Impact: Crops per decitonnes 1)	**Value ECU_{1991} (if not stated differently)**	**Value $€_{2000}$ before NewExt**	**Value $€_{2000}$ NewExt**
Barley – yield loss	5.4	6.3	6.3
Oats – yield loss	5.6	6.6	6.6
Potato – yield loss	8.2	9.6	9.6
Rice – yield loss	$\$_{1992}$ 274.4	254.9	254.9
Rye – yield loss	15.6	18.3	18.3
Sugar beet – yield loss	4.8	6.6	6.6
Sunflower seed – yield loss	ECU_{1994} 23.5	25.8	25.8
Tobacco – yield loss	nq	3414	3414
Wheat – yield loss	9.6	11.3	11.3
Fertiliser	ECU_{1990} 43	53	53
Lime	ECU_{1993} 1.7	1.8	1.8

1) please note, that the monetary values for crops will be evaluated and adapted in the ExternE-POL project. As the share of crop loss on overall external costs is low, this will however not have a substantial influence on the total external costs.

Impact: Material (per m^2)	**Value ECU_{1990}**	**Value $€_{2000}$ before NewExt**	**Value $€_{2000}$ NewExt**
Galvanised steel	ca. 30	country specific (14 – 45)	country specific (14 – 45)
Limestone	245	299	299
Mortar	27	33	33
Natural stone	245	299	299
Paint	11	13	13
Rendering	27	33	33
Sandstone	245	299	299
Zinc	22	27	27

4 Other methodological changes

Several changes in methodology have been made. In the following the major changes are listed:

- For the National Implementation calculations (1998 ExternE) background emission data were based on values from 1990. Now background emission data from 1998 (EMEP and Corinair) are used.
- The underlying grid has changed from Eurogrid (100 km x 100 km) to EMEP50 grid (50 km x 50 km).
- The meteorological data was updated with data from 1998 and it was adjusted to the new EMEP50 grid.
- Valuation of external costs due to CO_{2equiv} is now based on the IPCC (2001) global warming potential – 100 years period. Also the weighting factors have slightly changed. The weighting factors used in NewExt and National Implementation are shown in Table 9.

Table 9: Characteristic factors used in this study for calculation of the CO_{2equiv}

Greenhouse gases (relevant for power plants)	Characteristic factors NewExt (IPCC 2001)	National Implementation (IPCC 1995)
	kg CO_{2equiv} / kg	kg CO_{2equiv} / kg
CO_2	1	1
CH_4	23	21
N_2O	296	310

Numerous studies have sought to quantify the benefits of reducing emissions of greenhouse gases. One of the best examples was carried out within the ExternE Project (see (European Commission 1999c)). Externalities of greenhouse gas emissions have a wide range, of the order €5 to more than €100 / t CO_{2equiv} . For National Implementation, a range of values from 18 to 46 € per t of CO_{2equiv} was used based on a damage cost approach. For NewExt, a value of €19 / t CO_{2equiv} has been taken (see chapter II of the NewExt final report).

The emissions caused by the up- and downstream processes, such as fuel extraction, storage, transportation, refining, etc., of the different fuel cycles have also been assessed. These emissions accrue at several different locations in Europe and even outside of Europe (e.g. emissions caused by fuel extraction). In order to estimate the external costs, these emissions are multiplied with average damage factors. The damage factors are displayed in Table 20 and Table 21 of this report.

In the following, the impact of the changes in grid size and background concentration are analyzed:

Instead of the EUROGRID (100 x 100 km^2) with EMEP background emissions, the EMEP50 grid (50 x 50 km^2) with CORINAIR 90/94 background emissions is used in the new EcoSense version. The EMEP50 grid covers a larger area with 19% more population than the EUROGRID. The different background emission scenario leads to changes in air chemistry and therefore in changes in nitrate and sulphate damages.

Based on the identical ERFs and the identical monetary values a comparison of the results gained with the old and the new EcoSense model of regional, local and total impacts (sum of regional and local results, corrected by the share of regional damages within the local area) for a coal fired power station in Lauffen (Germany) is displayed in Table 10 (new model divided by old model):

Table 10: Ratio of impacts quantified with new (NewExt) and old (National Implementation) EcoSense version, using the same monetary values and same dose response functions [ratio: new / old]

	Local	**Regional**	**LocReg**
Nitrates	nq	86%	nq
Sulphates	nq	135%	nq
PM_{10}	105%	134%	100%
SO_2	105%	148%	115%

This means that the new results for nitrates emissions are by a factor of 0,86 smaller than former results and so on.

Especially the result for nitrates is interesting because they created more than 60 % of the external costs of the coal fired power station in Lauffen/Germany.

5 Data Input

The input data for the different technologies in this chapter are taken from the corresponding National Implementation reports, which can be found under (ExternE Homepage). Theses emissions do not represent the state of the art at the time writing this report. The data is from the mid nineties. Improvements in plant technology have reduced emissions significantly. A good example are the power plants in Belgium. At the beginning of the National Implementation project emissions were taken from existing power plants, or were based on plans and literature for plants with flue gas cleaning. Now the flue gas cleaning has been implemented and working, and has proven to be more effective for SO_2 and particles, than thought previously. Reduction percentages for SO_2 are well above 90% and about 75% of particles is retained in the wet gas cleaning. For gas new combined cycle gas turbines (CCGT's) are operating with an efficiency of 55%. Other coal fired power plants are still comparable to the case A in Table 11.

However, in order to indicate the influence on results of impact assessment due to the updated methodology the same data has to be used for new calculations as was used for the National Implementation reports. Likewise, for the up- and downstream processes data is taken from the National Implementation reports. In the National Implementation reports there is no exact information available about the location of the emissions caused by up- and downstream processes. In case of conventional technologies, i.e. gas or coal fired power stations the up- and downstream processes have only a relatively small influence toward the final result. Therefore, it is necessary and sufficient to use the damage factors displayed in Table 20 and Table 21 in order to estimate the impacts of these emissions. For low emissions the damage factors for PM_{10} and SO_2 include a surplus for local damages (the approach, how to estimate local damages for multi-source emissions is described in (European Commission 2003a)) When the emissions are low, i.e. lower than 100 meter a large quantity of the damages due to primary particles arise in the local area. The emission height of the up- and downstream processes is not indicated in the National Implementation reports and therefore, the applied damage factors are the average of damage factors for low and high emissions.

An example for the application of the impact assessment methodology after NewExt with regard to new updated emission data is carried out for the power generation in France. The new emission data from EdF are shown in Table 15.

5.1 Belgium

5.1.1 Power generation Belgium

Table 11: Input data for power generation in Belgium

		Hard Coal A	Hard Coal B	Gas
Location		**Genk-Langerlo**	**Genk-Langerlo**	**Drogenbos**
Plant type		**Case A (no FGD nor SCR)**	**Case B (with FGD and SCR)**	**Combined cycle gas turbine**
Generator capacity	MW	300	300	467
Electricity sent out	MW	274	266	460
Net efficiency	%	38	37	51
Annual generation	GWh	1517	1472	2433
Data relevant for atmospheric transport modeling				
Stack height	m	140	140	60
Stack diameter	m	5	5	5
Flue gas volume stream (full load)	Nm3/h	1.02E+06	1.02E+06	2.80E+06
Flue gas temperature	K	387	387	493
Latitude	degree	50.97	50.97	50.78
Longitude	degree	5.50	5.50	4.17
Elevation at Site	m	100	100	100
Emissions caused by electricity generation				
SO_2	mg/kWh	4,490	460	1
NO_x	mg/kWh	3,800	790	270
Particulates	mg/kWh	130	80	0
CO_2	g/kWh	889	920	nd
CH_4	mg/kWh	nd	nd	nd
N_2O	mg/kWh	nd	nd	nd
CO_{2equiv}	g/kWh	889	920	387

5.1.2 Up- and downstream processes Belgium

Emissions caused by up- and downstream processes, such as fuel extraction, storage, transportation, refining, etc. are shown in Table 12.

Table 12: Emissions caused by up- and downstream processes - Belgium

		Hard Coal A	Hard Coal B	Gas
SO_2	mg/kWh	760	760	nd
NO_x	mg/kWh	530	530	nd
Particulates	mg/kWh	40	40	nd
CO_{2equiv}	g/kWh	47	47	22

5.2 France

For France data from National Implementation and new data from the EdF (EdF 2003) have been available.

5.2.1 Power generation France - Data from National Implementation (NI)

Table 13: Input data for power generation in France - Data from National Implementation (NI)

		France (NI)	France (NI)	France (NI)
		Hard Coal	**Oil**	**Gas**
Location		Cordemais, near Nantes	Cordemais, near Nantes	Cordemais, near Nantes
Plant type		Existing plant, pulverised fuel, hypothetical installation of flue gas desulfurisation, steam turbine	Existing plant, low S oil, steam turbine	Hypothetical new plant, gas turbine combined cycle
Generator capacity	MW	600	700	250
Electricity sent out	MW	600	700	250
Net efficiency	%	38	39	52
Annual generation	GWh	2100	1050	1500
Data relevant for atmospheric transport modeling				
Stack height	m	220	150	110
Stack diameter	m	10	10	10
Flue gas volume stream (full load)	Nm3/h	2.77E+06	2.77E+06	2.77E+06
Flue gas temperature	K	500	500	500
Latitude	degree	47.18	47.18	47.18
Longitude	degree	-1.48	-1.48	-1.48
Elevation at Site	m	100	100	100
Emissions caused by electricity generation				
SO_2	mg/kWh	1,360	5,260	ng
NO_x	mg/kWh	2,220	1,200	710
Particulates	mg/kWh	170	130	ng
VOC	mg/kWh	45	480	24
CO_2	g/kWh	900	740	401
CH_4	mg/kWh	nd	nd	nd
N_2O	mg/kWh	nd	nd	nd
CO_{2equiv}	g/kWh	1,085	866	433

5.2.2 Up- and downstream processes France - Data from National Implementation

Table 14: Emissions caused by up- and downstream processes, such as fuel extraction, storage, transportation, refining, etc. France - Data from National Implementation (NI)

		France (NI)	France (NI)	France (NI)
		Hard Coal	**Oil**	**Gas**
SO_2	mg/kWh	489	1565	60
NO_x	mg/kWh	68	80	150
Particulates	mg/kWh	228	nd	Nd
VOC	mg/kWh	4300	370	nd
CO_{2equiv}	g/kWh	134	93	178

5.2.3 Power generation France - New Data from EdF

Table 15: Input data for power generation in France - New Data from EdF

		France (New)	France (New)	France (New)
		Hard Coal	**Oil**	**Gas**
Location		**Cordemais boiler n°4, near Nantes**	**Cordemais boiler n°2, near Nantes**	**Cordemais, near Nantes**
Plant type		Existing plant, pulverised coal, flue gas desulfurisation (actually equipped), steam turbine	Existing plant Low S oil, steam turbine	Hypothetical new plant, gas turbine combined cycle
Generator capacity	MW	Nd	nd	nd
Electricity sent out	MW	600	700	400
Net efficiency	%	Nd	nd	55
Annual generation	GWh	2700	350	1800
Data relevant for atmospheric transport modeling				
Stack height	m	220	149	150
Stack diameter	m	2x3.6	6	6
Flue gas volume stream (full load)	Nm3/h	1.96E+06	2.15E+06	1.80E+06
Flue gas temperature	K	363	433	383
Latitude	degree	47.18	47.18	47.18
Longitude	degree	-1.48	-1.48	-1.48
Elevation at Site	m	100	100	100
Emissions caused by electricity generation				
SO_2	mg/kWh	824	4,321	7
NO_x	mg/kWh	2,678	2,113	225
Particulates	mg/kWh	7	129	0
NMVOC	mg/kWh	13	26	1
CO_2	g/kWh	810	673	370
CH_4	mg/kWh	10	2	15
N_2O	mg/kWh	22	22	0
CO_{2equiv}	g/kWh	817	680	370

5.3 Germany

5.3.1 Power generation Germany

Table 16: Input data for power generation in Germany

		Hard Coal	Oil	Gas
Location		**Lauffen**	**Lauffen**	**Lauffen**
Plant type		Pulverised coal power plant with FGD, DENOX, and dedusting	Gas-turbine peak load power plant	Combined cycle
Generator capacity	MW	652	157	791
Electricity sent out	MW	600	156	778
Net efficiency	%	43	31	58
Annual generation	GWh	3900	105	5054
Data relevant for atmospheric transport modeling				
Stack height	m	240	170	250
Stack diameter	m	10	6	10
Flue gas volume stream (full load)	Nm3/h	1.72E+06	1.43E+06	3.23E+06
Flue gas temperature	K	403	433	364
Latitude	degree	49.08	49.08	49.08
Longitude	degree	9.18	9.18	9.18
Elevation at Site	m	165	165	165
Emissions caused by electricity generation				
SO_2	mg/kWh	288	1,088	0
NO_x	mg/kWh	516	814	208
Particulates	mg/kWh	57	18	0
CO_2	g/kWh	781	858	348
CH_4	mg/kWh	42	35	27
N_2O	mg/kWh	42	60	1
CO_{2equiv}	g/kWh	794	877	349

5.3.2 Up- and downstream processes Germany

Table 17: Emissions caused by up- and downstream processes, such as fuel extraction, storage, transportation, refining, etc. Germany

		Hard Coal	Oil	Gas
SO_2	mg/kWh	38	404	3
NO_x	mg/kWh	44	171	69
Particulates	mg/kWh	125	49	18
CO_{2equiv}	g/kWh	110	80	53

5.4 United Kingdom

5.4.1 Power generation United Kingdom

Table 18: Input data for power generation in the United Kingdom

		Hard Coal	Oil	Gas
Location		**West Burton**	**Fawely, Hampshire (south coast)**	**West Burton**
Plant type		Coal-fired station with FGD	Combined cycle oil-fired power station	Combined cycle gas turbine (CCGT)
Generator capacity	MW	1800	548	652
Electricity sent out	MW	1800	528	652
Net efficiency	%	38	48	52
Annual generation	GWh	11700	3431	4238
Data relevant for atmospheric transport modeling				
Stack height	m	230	250	65
Stack diameter	m	10	10	10
Flue gas volume stream (full load)	Nm3/h	1.77E+06	3.76E+06	2.03E+06
Flue gas temperature	K	428	428	373
Latitude	degree	53.38	50.90	53.38
Longitude	degree	-1.50	-1.38	-1.50
Elevation at Site	m	100	20	100
Anemometer Height	m	10	10	10
Emissions caused by electricity generation				
SO_2	mg/kWh	1,100	798	nd
NO_x	mg/kWh	2,200	798	460
Particulates	mg/kWh	160	12	nd
CO_2	g/kWh	855	608	393
CH_4	mg/kWh	nd	23	nd
N_2O	mg/kWh	60	15	13
CO_{2equiv}	g/kWh	873	613	397

5.4.2 Up- and downstream processes in the United Kingdom

Table 19: Emissions caused by up- and downstream processes, such as fuel extraction, storage, transportation, refining, etc.

		UK	UK	UK
		Hard Coal	Oil	Gas
SO_2	mg/kWh	ng	228	ng
NO_x	mg/kWh	nq	190	ng
Particulates	mg/kWh	ng	7	ng
VOC	mg/kWh	nd	660	nd
CO_{2equiv}	g/kWh	87	49	13

6 Summary of the Results

In the following the results of the calculations using EcoSense are summarized. With the input data listed in chapter 5 calculations have been made using the state of methodology at National Implementation, before and of NewExt.

6.1 Damage factors from ExternE projects before NewExt and from NewExt

Damage factors are used to evaluate the external costs caused by up – and downstream emissions. In Table 20 and Table 21 damage factors for the average of high stack and low stack emissions within the EU15 are displayed. The pollutants NO_x, SO_2, and NMVOC have an impact on human health (HH), crops and materials. For PM_{10} only the impact on human health is evaluated. The damages due to PM_{10} occur mainly in the local area if the emission height, e.g. stack height, is low. Therefore, according to (European Commission 2003a), a surplus of 24,043 €/t (before NewExt) has to be added to the regional damages caused by PM_{10} in the case of low stack emissions. After NewExt this value increased to 32,737 €/t. A part of the damages caused by SO_2, i.e. the damages to human health due to SO_2 also need a surplus for local damages of low emissions. These are 1,008 €/t before and 466 €/t after NewExt. In contrast to the increase of the value for PM_{10} the value for SO_2 decreases because the monetary valuation of acute mortality (VLYL) decreased from 165.700 € to 75.000 € while the ERF for acute mortality due to SO_2 does not change. Since the emission height of the up- and downstream processes is not indicated in the National Implementation reports the results for the "Other fuel chain stages" in Table 22 to Table 27 show the up- and downstream processes assuming an average of low and high release of the emissions. The results are subdivided in external costs due to *power generation* and *other fuel chain stages*. The external costs of the power generation are again subdivided into human health impacts, global warming and others (in this case others comprises crops and building materials). The external costs of the other fuel chain stages are subdivided into global warming and others (in this case others comprises crops, building materials and human health). These distinction and presentation of results was used in the National Implementation reports and hence, had to be applied towards the new results in order to enable comparison of the results.

Note: Table 20 and Table 21 show negative marginal costs, as additional emissions of NO (nitrogen monoxide) currently reduces the ozone concentration near the source, while it may lead to an increase of ozone farer away. If the population density is high near the source, this leads to a net reduction of exposure to ozone and thus to an external benefit.

However, substantial reductions of emissions of nitrogen oxides and volatile organic compounds on a regional scale could lead to still lower ozone level and lower external costs of total air pollution. Increasing NO_x emissions would thus increase the costs to reach the optimal reduction strategy, on the other hand a reduction of NO_x emissions would be a step *towards reaching the optimal reduction. That a reduction of NO_x emission could be beneficial as part of a wider ozone reduction strategy is not reflected in the negative figures given above.*

Table 20: Damage factors before NewExt [€/t]

[€/t]	EU15 (average of low and high stacks)
NO_x Total	2,084
NO_x Crops and Materials	195
NO_x ozone human health (HH) (see note above)	-399
NO_x not ozone HH	2,288
SO_2 Total	4,268
PM_{10}	19,872
NMVOC Total (Crops and HH O_3)	1,215
NMVOC (Crops O_3)	644
NMVOC (HH O_3)	571

Table 21: Damage factors from NewExt [€/t]

[€/t]	EU15 (average of low and high stacks)
NO_x Total	3,021
NO_x Crops and Materials	195
NO_x ozone human health (HH) (see note above)	-335
NO_x not ozone HH	3,161
SO_2 Total	3,524
PM_{10}	27,042
NMVOC Total (Crops and HH O_3)	1,124
NMVOC (Crops O_3)	644
NMVOC (HH O_3)	480

6.2 Results for the different fuel cycles before and of NewExt

Table 22: Results of the coal fuel cycle before NewExt [€-Cent/kWh]

	Site, size [MW]	Technology	Power generation: Human health	Power generation: Global warming	Power generation: Other	Other fuel chain stages: Global warming	Other fuel chain stages: Other	**Sub-Total**
Be	Genk, 300	No FGD nor SCR	4.50	1.69	-0.27	0.09	0.51	**6.51**
Be	Genk, 300	With FGD and SCR	0.61	1.75	-0.05	0.09	0.51	**2.90**
Fr	Cordemais, 600	Pulverized fuel, hypothetical FGD, steam turbine	1.62	2.06	0.11	0.25	0.68	**4.72**
Fr	Cordemais, 600 (new data)	Pulverized fuel, FGD (actual), steam turbine	1.39	1.55	0.13	nd	nd	**3.07**
Ge	Lauffen, 652	Pulverized fuel, FGD, DENOX, and dedusting	0.47	1.51	0.02	0.21	0.27	**2.47**
UK	West Burton, 1800	Coal-fired station with FGD	0.67	1.66	-0.08	0.16	nd	**2.42**

Table 23: Results of the coal fuel cycle from NewExt [€-Cent/kWh]

	Site, size [MW]	Technology	Power generation: Human health	Power generation: Global warming	Power generation: Other	Other fuel chain stages: Global warming	Other fuel chain stages: Other	**Sub-Total**
Be	Genk, 300	No FGD nor SCR	4.07	1.69	-0.07	0.09	0.54	**6.33**
Be	Genk, 300	With FGD and SCR	0.65	1.75	-0.03	0.09	0.54	**3.00**
Fr	Cordemais, 600	Pulverized fuel, hypothetical FGD, steam turbine	1.77	2.06	0.14	0.25	0.81	**5.03**
Fr	Cordemais, 600 (new data)	Pulverized fuel, FGD (actual), steam turbine	1.63	1.55	0.16	nd	nd	**3.34**
Ge	Lauffen, 652	Pulverized fuel, FGD, DENOX, and dedusting	0.51	1.51	0.02	0.21	0.36	**2.61**
UK	West Burton, 1800	Coal-fired station with FGD	0.75	1.66	-0.05	0.16	nd	**2.53**

Table 24: Results of the oil fuel cycle before NewExt [€-Cent/kWh]

	Site, size [MW]	Technology	Power generation: Human health	Power generation: Global warming	Power generation: Other	Other fuel chain stages: Global warming	Other fuel chain stages: Other	**Sub-Total**
Fr	Cordemais, 700	Low S oil, steam turbine	3.42	1.65	0.04	0.18	0.68	**5.97**
Fr	Cordemais, 700 (new data)	Low S oil, steam turbine	3.22	1.29	0.09	nd	nd	**4.60**
Ge	Lauffen, 157	Gas-turbine peak load power plant	1.13	1.67	0.05	0.15	0.36	**3.36**
UK	Fawely, Hampshire (south coast), 528	Combined cycle oil-fired power station	0.74	1.16	-0.03	0.09	0.15	**2.11**

Table 25: Results of the oil fuel cycle: NewExt-methodology [€-Cent/kWh]

	Site, size [MW]	Technology	Power generation: Human health	Power generation: Global warming	Power generation: Other	Other fuel chain stages: Global warming	Other fuel chain stages: Other	**Sub-Total**
Fr	Cordemais, 700	Low S oil, steam turbine	2.98	1.65	0.12	0.18	0.58	**5.50**
Fr	Cordemais, 700 (new data)	Low S oil, steam turbine	3.00	1.29	0.16	nd	nd	**4.45**
Ge	Lauffen, 157	Gas-turbine peak load power plant	1.11	1.67	0.05	0.15	0.33	**3.30**
UK	Fawely, Hampshire (south coast), 528	Combined cycle oil-fired power station	0.73	1.16	-0.01	0.09	0.16	**2.14**

Table 26: Results of the gas fuel cycle before NewExt [€-Cent/kWh]

	Site, size [MW]	Technology	Power generation: Human health	Power generation: Global warming	Power generation: Other	Other fuel chain stages: Global warming	Other fuel chain stages: Other	**Sub-Total**
Be	Drogenbos, 467	Combined cycle gas turbine	0.05	0.74	-0.02	0.04	nd	**0.81**
Fr	Cordemais, 250	Hypothetical new plant, combined cycle gas turbine	0.25	0.76	0.04	0.34	0.06	**1.45**
Fr	Cordemais, 400 (new data)	Hypothetical new plant, combined cycle gas turbine	0.09	0.7	0.01	nd	nd	**0.80**
Ge	Lauffen, 791	Combined cycle	0.10	0.66	0.00	0.1	0.05	**0.91**
UK	West Burton, 652	Combined cycle gas turbine (CCGT)	0.01	0.75	-0.02	0.02	nd	**0.77**

Table 27: Results of the gas fuel cycle: NewExt-methodology [€-Cent/kWh]

	Site, size [MW]	Technology	Power generation: Human health	Power generation: Global warming	Power generation: Other	Other fuel chain stages: Global warming	Other fuel chain stages: Other	**Sub-Total**
Be	Drogenbos, 467	Combined cycle gas turbine	0.09	0.74	-0.02	0.04	nd	**0.85**
Fr	Cordemais, 250	Hypothetical new plant, combined cycle gas turbine	0.34	0.76	0.04	0.34	0.07	**1.55**
Fr	Cordemais, 400 (new data)	Hypothetical new plant, combined cycle gas turbine	0.11	0.7	0.01	nd	nd	**0.83**
Ge	Lauffen, 791	Combined cycle	0.09	0.66	0.00	0.1	0.07	**0.93**
UK	West Burton, 652	Combined cycle gas turbine (CCGT)	0.05	0.75	-0.01	0.02	nd	**0.80**

In addition to these detailed results shown in Table 22 to Table 27, the following Table 28 shows the overall comparison of the subtotal results gained in the ExternE National Implementation phase (European Commission 1999a) and those with all the updates since then including the NewExt methodology. Here, the step in between, i.e. the "before NewExt" status used in the GREENSENSE project, where parts of the changes described above have already been realized (see the description in the following Chapter 7), has been omitted.

Table 28: Results of the coal, oil and natural gas fuel cycles, ExternE National Implementation (1999a) and NewExt methodology (2004) [€-Cent/kWh]

	Site, size [MW]	Technology	**Subtotal National Implementation (1999a)** 1)	**Subtotal NewExt (2004)**
Coal Fuel Cycle				
Be	Genk, 300	No FGD nor SCR	**12.3**	**6.33**
Be	Genk, 300	With FGD and SCR	**3.7**	**3.00**
Fr	Cordemais, 600	Pulverized fuel, FGD (hypothetical), steam turbine	**6.9**	**5.03**
Fr	Cordemais, 600 (new data)	Pulverized fuel, FGD (actually installed), steam turbine	nd	**3.34**
Ge	Lauffen, 652	Pulverized fuel, FGD, DENOX, and dedusting	**3.0**	**2.61**
UK	West Burton, 1800	Coal-fired station with FGD	**4.2**	**2.53**
Oil Fuel Cycle				
Fr	Cordemais, 700	Low S oil, steam turbine	**8.4**	**5.50**
Fr	Cordemais, 700 (new data)	Low S oil, steam turbine	nd	**4.45**
Ge	Lauffen, 157	Gas-turbine peak load power plant	**5.1**	**3.30**
UK	Fawely, Hampshire (south coast), 528	Combined cycle oil-fired power station	**3.3**	**2.14**
Natural Gas Fuel Cycle				
Be	Drogenbos, 467	Combined cycle gas turbine	**1.1**	**0.85**
Fr	Cordemais, 250	Hypothetical new plant, combined cycle gas turbine	**1.9**	**1.55**
Fr	Cordemais, 400 (new data)	Hypothetical new plant, combined cycle gas turbine	nd	**0.83**
Ge	Lauffen, 791	Combined cycle	**1.2**	**0.93**
UK	West Burton, 652	Combined cycle gas turbine (CCGT)	**1.1**	**0.80**

1) National implementation results included occupational health, which was not taken into respect in later ExternE phases. For global warming damages, mid values of damage costs for an underlying discount rate of 3 % have been used.

7 Detailed Results using the 'National Implementation', 'Before NewExt' and 'NewExt' methodologies

7.1 Results for power generation and up- and downstream processes from National Implementation (NI) and results before NewExt

Results for greenhouses gases (GHG), i.e. global warming from National Implementation reports are based on a mid estimate using a discount rate of 3 % (European Commission 1999b). The results for NewExt are based on an valuation of 19 €$_{2000}$ per tonne of $CO_{2equiv.}$

7.1.1 Belgium

Table 29: Power generation Belgium NI (national implementation) and before NewExt - Belgium

[Euro-Cent / kWh]	Hard Coal A		Hard Coal B		Gas	
	before NewExt	**NI**	**before NewExt**	**NI**	**before NewExt**	**NI**
Public health						
Mortality - YOLL	3.51	8.78	0.49	1.53	0.05	0.25
of which TSP	0.13	0.20	0.08	0.22	0.00	0.00
SO_2 as sulphates	2.71	nd	0.27	nd	0.00	0.00
SO_2 as SO_2	0.30	nd	0.03	nd	0.00	0.00
SO_2 as total	3.01	4.63	0.30	0.46	0.00	0.001
NO_x as total	0.37	3.77	0.10	0.83	0.05	0.24
NO_x (via ozone)	-0.15	0.15	-0.03	0.03	-0.01	0.01
NO_x (via nitrates)	0.52	nd	0.14	nd	0.06	nd
Morbidity	0.99	1.34	0.12	0.24	0.00	0.05
of which TSP, SO_2, NOx	1.35	1.07	0.20	0.19	0.02	0.03
NO_x (via ozone)	-0.36	0.27	-0.08	0.06	-0.03	0.02
Crops	-0.27	0.13	-0.05	0.03	-0.02	nd
of which SO_2	-0.03	0.004	-0.003	0.002	0.00	0.00
NO_x (via acid and N dep.)	-0.003	nq	-0.001	nq	-0.001	nq
NO_x (via ozone)	-0.24	0.13	-0.05	0.03	-0.02	0.01
Materials	0.21	0.22	0.03	0.004	0.003	0.003
CO_{2equiv}	1.69	1.6	1.75	1.66	0.74	0.70

Table 30: Up- and Downstream processes NI and before NewExt - Belgium

[Euro-Cent / kWh]	Hard Coal A		Hard Coal B		Gas	
	before NewExt	NI	before NewExt	NI	before NewExt	NI
SO_2	0.32	nq	0.32	nq	nd	nq
NO_x	0.11	nq	0.11	nq	nd	nq
Particulates	0.08	nq	0.08	nq	nd	nq
Sum air pollutants	0.51	0.02	0.51	0.02	nd	nd
CO_{2equiv}	0.09	0.14	0.09	0.14	0.04	0.04
Sum air pollutants and CO_{2equiv}	0.60	0.16	0.60	0.16	nd	nd

7.1.2 France

Table 31: Power generation NI and before NewExt - France

[Euro-Cent / kWh]	Hard Coal			Oil			Gas		
	before NewExt NI data	before NewExt EdF data	NI	before NewExt NI data	before NewExt EdF data	NI	before NewExt NI data	before NewExt EdF data	NI
Public health									
Mortality - YOLL	1.16	0.99	4.02	2.47	2.32	5.71	0.18	0.06	0.92
of which TSP	0.06	0.00	0.09	0.05	0.05	0.07	0.00	0.00	ng
SO_2 as sulphates	0.53	0.32	nq	2.05	1.68	nq	0.00	0.00	nq
SO_2 as SO_2	0.04	0.02	nq	0.14	0.11	nq	0.00	0.00	nq
SO_2 as total	0.56	0.34	1.05	2.18	1.80	4.07	0.00	0.00	ng
NO_x as total	0.53	0.65	2.77	0.24	0.48	1.5	0.18	0.06	0.88
NO_x (via ozone)	0.01	0.01	0.11	0.00	0.00	0.06	0.00	0.00	0.03
NO_x (via nitrates)	0.53	0.64	nq	0.24	0.47	nq	0.17	0.06	nq
NMVOC (via ozone)	ng	ng	0.001	ng	ng	0.01	ng	ng	0.001
Morbidity	0.46	0.40	0.82	0.94	0.90	1.06	0.07	0.03	0.21
of which TSP, SO_2, NO_x	0.45	0.39	0.66	0.94	0.88	0.95	0.07	0.02	0.16
NO_x (via ozone)	0.01	0.01	0.16	0.01	0.01	0.09	0.00	0.00	0.05
NMVOC (via ozone)	0.00	0.00	0.002	0.00	0.00	0.02	ng	0.00	0.001
Crops	0.11	0.13	0.07	0.03	0.09	0.05	0.04	0.01	0.02
of which SO_2	-0.01	-0.01	0.001	-0.02	-0.02	0.002	0.000	0.000	ng
NO_x (via acid and N dep.)	-0.01	-0.01	nq	0.01	0.000	nq	-0.003	-0.001	nq
NO_x (via ozone)	0.12	0.15	0.07	0.05	0.12	0.04	0.039	0.01	0.02
NMVOC (via ozone)	0.00	0.00	0.001	0.03	0.00	0.01	0.00	0.00	0.001
Materials	0.03	0.03	0.01	0.05	0.07	0.05	0.001	0.001	ng
CO_{2equiv}	2.06	1.55	1.62	1.65	1.29	1.33	0.76	0.70	0.72

Table 32: Up- and Downstream processes NI and before NewExt - France, only NI data

[Euro-Cent / kWh]	Hard Coal		Oil		Gas	
	before NewExt	NI	before NewExt	NI	before NewExt	NI
SO_2	0.21	nq	0.67	nq	0.03	nq
NO_x	0.01	nq	0.02	nq	0.03	nq
Particulates	0.45	nq	nd	nd	nd	nd
Sum air pollutants	0.68	0.02	0.68	0.02	0.06	nq
CO_{2equiv}	0.25	0.14	0.18	0.14	0.34	0.06
Sum air pollutants and CO_{2equiv}	0.93	0.16	0.86	0.16	0.39	nd

7.1.3 Germany

Table 33: Power generation NI and before NewExt - Germany

[Euro-Cent / kWh]	Hard Coal		Oil		Gas	
	before NewExt	**NI**	**before NewExt**	**NI**	**before NewExt**	**NI**
Public health						
Mortality - YOLL	0.35	1.04	0.90	2.26	0.05	0.24
of which TSP	0.05	0.11	0.02	0.04	nd	nq
SO_2 as sulphates	0.17	nq	0.66	nq	nd	nq
SO_2 as SO_2	0.02	nq	0.07	nq	nd	nq
SO_2 as total	0.19	0.29	0.73	1.29	nd	nd
NO_x as total	0.11	0.63	0.15	0.92	0.05	0.24
NO_x (via ozone)	-0.01	0.01	-0.01	0.02	0.00	0.01
NO_x (via nitrates)	0.12	nq	0.17	nq	0.05	nq
Morbidity	0.12	0.15	0.23	0.31	0.05	0.04
of which TSP, SO_2, NO_x	0.14	0.13	0.25	0.28	0.07	0.03
NO_x (via ozone)	-0.02	0.02	-0.02	0.03	-0.02	0.01
Crops	-0.003	0.001	-0.01	0.002	0.00	0.00
of which SO_2	-0.002	0.00	-0.01	0.001	0.00	0.00
NO_x (via acid and N dep.)	-0.001	-0.00	-0.001	-0.00	0.00	-0.00
NO_x (via ozone)	0.00	0.001	0.00	0.002	0.00	0.00
Materials	0.02	0.01	0.05	0.04	0.003	0.003
CO_{2equiv}	1.51	1.43	1.67	1.56	0.66	0.63

Table 34: Up- and Downstream processes NI and before NewExt - Germany

[Euro-Cent / kWh]	Hard Coal		Oil		Gas	
	before NewExt	NI	before NewExt	NI	before NewExt	NI
SO_2	0.02	nq	0.17	nq	0.001	nq
NO_x	0.01	nq	0.04	nq	0.01	nq
Particulates	0.25	nq	0.10	nq	0.04	nq
Sum air pollutants	0.27	0.15	0.31	0.78	0.05	0.15
CO_{2equiv}	0.21	0.19	0.15	0.14	0.10	0.09
Sum air pollutants and CO_{2equiv}	0.48	0.34	0.46	0.92	0.15	0.24

7.1.4 United Kingdom

Table 35: Power generation NI and before NewExt - United Kingdom

[Euro-Cent / kWh]	Hard Coal		Oil		Gas	
	before NewExt	**NI**	**before NewExt**	**NI**	**before NewExt**	**NI**
Public health						
Mortality - YOLL	0.59	1.95	0.58	1.41	0.03	0.26
of which TSP	0.10	0.20	0.01	0.02	nd	ng
SO_2 as sulphates	0.33	nq	0.38	nq	nd	nq
SO_2 as SO_2	0.04	nq	0.04	nq	nd	nq
SO_2 as total	0.37	0.61	0.42	0.68	nd	0.00
NO_x as total	0.12	1.05	0.15	0.68	0.03	0.24
NO_x (via ozone)	-0.07	0.09	-0.03	0.03	-0.01	0.02
NO_x (via nitrates)	0.19	nq	0.17	nq	0.05	nq
Morbidity	0.08	0.39	0.16	nq	-0.02	0.08
of which TSP, SO_2, NO_x	0.25	0.23	0.23	0.17	0.02	0.04
NO_x (via ozone)	-0.17	0.16	-0.06	0.06	-0.04	0.03
Crops	-0.07	0.08	-0.03	0.03	-0.02	0.02
of which SO_2	-0.002	0.002	-0.004	0.001	0.00	0.00
NO_x (via acid and N dep.)	-0.003	nq	-0.001	nq	-0.001	nq
NO_x (via ozone)	-0.07	0.08	-0.03	0.03	-0.01	0.02
Materials	0.03	0.07	0.02	0.04	0.002	0.003
CO_{2equiv}	1.66	1.50	1.16	1.09	0.75	0.71

Table 36: Up- and Downstream processes NI and before NewExt - United Kingdom

[Euro-Cent / kWh]	Hard Coal		Oil		Gas	
	before NewExt	NI	before NewExt	NI	before NewExt	NI
SO_2	nd	nq	0.10	nq	nd	nq
NO_x	nd	nq	0.04	nq	nd	nq
Particulates	nd	nq	0.01	nq	nd	nq
Sum air pollutants	nd	0.002	0.15	0.35	nd	nd
CO_{2equiv}	0.16	0.1	0.09	0.08	0.02	0.02
Sum air pollutants and CO_{2equiv}	nd	0.10	0.24	0.43	nd	nd

7.2 Results for power generation and up- and downstream processes before NewExt and with NewExt methodology

7.2.1 Belgium

Table 37: Power generation before and with NewExt - Belgium

[Euro-Cent / kWh]	Hard Coal A		Hard Coal B		Gas	
	before NewExt	NewExt	before NewExt	NewExt	before NewExt	NewExt
Public health						
Mortality - YOLL	3.51	2.95	0.49	0.50	0.05	0.12
of which TSP	0.13	0.16	0.08	0.10	0.00	0.00
SO_2 as sulphates	2.71	2.09	0.27	0.22	0.00	-0.003
SO_2 as SO_2	0.30	0.13	0.03	0.01	0.00	0.00
SO_2 as total	3.01	2.22	0.30	0.23	0.00	-0.003
NO_x as total	0.37	0.57	0.10	0.16	0.05	0.12
NO_x (via ozone)	-0.15	-0.07	-0.03	-0.01	-0.01	-0.01
NO_x (via nitrates)	0.52	0.63	0.14	0.18	0.06	0.13
Morbidity	0.99	1.05	0.12	0.17	0.00	0.02
of which TSP, SO_2, NO_x	1.35	1.41	0.20	0.25	0.02	0.06
NO_x (via ozone)	-0.36	-0.36	-0.08	-0.08	-0.03	-0.04
Crops	-0.27	-0.27	-0.05	-0.06	-0.02	-0.02
of which SO_2	-0.03	-0.03	-0.003	-0.003	0.00	0.00
NO_x (via acid and N dep.)	-0.003	-0.003	-0.001	-0.001	-0.001	-0.001
NO_x (via ozone)	-0.24	-0.24	-0.05	-0.05	-0.02	-0.02
Materials	0.21	0.21	0.03	0.03	0.003	0.003
CO_{2equiv}	1.69	1.69	1.75	1.75	0.74	0.74

Table 38: Up- and Downstream processes before and with NewExt - Belgium

[Euro-Cent / kWh]	Hard Coal A		Hard Coal B		Gas	
	before NewExt	NewExt	before NewExt	NewExt	before NewExt	NewExt
SO_2	0.32	0.27	0.32	0.27	nd	nd
NO_x	0.11	0.16	0.11	0.16	nd	nd
Particulates	0.08	0.11	0.08	0.11	nd	nd
Sum air pollutants	0.51	0.54	0.51	0.54	nd	nd
CO_{2equiv}	0.09	0.09	0.09	0.09	0.04	0.04
Sum air pollutants and CO_{2equiv}	0.60	0.63	0.60	0.63	nd	nd

7.2.2 France

Table 39: Power generation before and with NewExt - France

[Euro-Cent / kWh]	Hard Coal				Oil			
	before NewExt NI data	before NewExt EdF data	NewExt NI data	NewExt EdF data	before NewExt NI data	before NewExt EdF data	NewExt NI data	NewExt EdF data
Public health								
Mortality - YOLL	1.16	0.99	1.18	1.09	2.47	2.32	2.02	2.02
of which TSP	0.06	0.00	0.08	0.00	0.05	0.05	0.06	0.06
SO_2 as sulphates	0.53	0.32	0.41	0.24	2.05	1.68	1.58	1.30
SO_2 as SO_2	0.04	0.02	0.02	0.01	0.14	0.11	0.06	0.05
SO_2 as total	0.56	0.34	0.42	0.25	2.18	1.80	1.64	1.35
NO_x as total	0.53	0.65	0.68	0.83	0.24	0.48	0.31	0.61
NO_x (via ozone)	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00
NO_x (via nitrates)	0.53	0.64	0.68	0.83	0.24	0.47	0.31	0.61
NMVOC (via ozone)	0.00	0.00	ng	ng	0.00	0.00	ng	ng
Morbidity	0.46	0.40	0.59	0.55	0.94	0.90	0.96	0.98
of which TSP, SO_2, NO_x	0.45	0.39	0.58	0.53	0.94	0.88	0.95	0.97
NO_x (via ozone)	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
NMVOC (via ozone)	ng	ng	ng	ng	ng	ng	ng	ng
Crops	0.11	0.13	0.11	0.13	0.03	0.093	0.03	0.09
of which SO_2	-0.01	-0.01	-0.01	0	-0.02	-0.02	-0.02	-0.02
NO_x (via acid and N dep.)	-0.01	-0.01	-0.01	-0.01	0.01	0	0.01	0
NO_x (via ozone)	0.12	0.15	0.12	0.15	0.05	0.12	0.05	0.12
NMVOC (via ozone)	0.00	0.00	0.00	0.00	0.03	0.00	0.03	0.00
Materials	0.03	0.03	0.03	0.03	0.05	0.07	0.05	0.07
CO_{2equiv}	2.06	1.55	2.06	1.55	1.65	1.29	1.65	1.29

[Euro-Cent / kWh]	Gas			
	before NewExt NI data	**before NewExt EdF data**	**NewExt NI data**	**NewExt EdF data**
Public health				
Mortality - YOLL	0.18	0.06	0.22	0.07
of which TSP	nd	nd	nd	nd
SO_2 as sulphates	nd	nd	nd	nd
SO_2 as SO_2	nd	nd	nd	nd
SO_2 as total	nd	nd	nd	nd
NO_x as total	0.18	0.06	0.22	0.07
NO_x (via ozone)	0.00	0.00	0.00	0.00
NO_x (via nitrates)	0.17	0.06	0.22	0.07
NMVOC (via ozone)	ng	ng	ng	ng
Morbidity	0.07	0.03	0.12	0.04
of which TSP, SO_2, NO_x	0.07	0.02	0.11	0.04
NO_x (via ozone)	0.00	0.00	0.00	0.00
NMVOC (via ozone)	ng	ng	ng	ng
Crops	0.04	0.01	0.04	0.01
of which SO_2	0.00	0.00	0.00	0.00
NO_x (via acid and N dep.)	-0.003	-0.001	-0.003	-0.001
NO_x (via ozone)	0.04	0.01	0.04	0.01
NMVOC (via ozone)	0.00	0.00	0.00	0.00
Materials	0.001	0.001	0.001	0.001
CO_{2equiv}	0.76	0.70	0.76	0.70

Table 40: Up- and Downstream processes before and after NewExt - France

[Euro-Cent / kWh]	Hard Coal		Oil		Gas	
	before NewExt	NewExt	before NewExt	NewExt	before NewExt	NewExt
SO_2	0.21	0.17	0.67	0.55	0.03	0.02
NO_x	0.01	0.02	0.02	0.02	0.03	0.05
Particulates	0.45	0.62	nd	nd	nd	nd
Sum air pollutants	0.68	0.81	0.68	0.58	0.06	0.07
CO_{2equiv}	0.25	0.25	0.18	0.18	0.34	0.34
Sum air pollutants and CO_{2equiv}	0.93	1.06	0.86	0.75	0.39	0.40

7.2.3 Germany

Table 41: Power generation before and after NewExt – Germany

[Euro-Cent / kWh]	Hard Coal		Oil		Gas	
	before NewExt	**NewExt**	**before NewExt**	**NewExt**	**before NewExt**	**NewExt**
Public health						
Mortality - YOLL	0.35	0.36	0.90	0.77	0.05	0.07
of which TSP	0.05	0.07	0.02	0.02	0.00	0.00
SO_2 as sulphates	0.17	0.13	0.66	0.51	0.00	0.00
SO_2 as SO_2	0.02	0.01	0.07	0.03	0.00	0.00
SO2 as total	0.19	0.14	0.73	0.54	0.00	0.00
NO_x as total	0.11	0.15	0.15	0.21	0.05	0.07
NO_x (via ozone)	-0.01	0.00	-0.01	-0.01	0.00	0.00
NO_x (via nitrates)	0.12	0.16	0.17	0.22	0.05	0.07
Morbidity	0.12	0.15	0.23	0.33	0.05	0.03
of which TSP, SO_2, NO_x	0.14	0.18	0.25	0.37	0.07	0.03
NO_x (via ozone)	-0.02	-0.02	-0.02	-0.04	-0.02	-0.01
Crops	-0.003	-0.003	-0.01	-0.01	0.00	0.00
of which SO_2	-0.002	-0.002	-0.01	-0.01	0.00	0.00
NO_x (via acid and N dep.)	-0.001	-0.001	-0.001	-0.001	0.00	0.00
NO_x (via ozone)	0.00	0.00	0.00	0.00	0.00	0.00
Materials	0.02	0.02	0.05	0.05	0.003	0.003
CO_{2equiv}	1.51	1.51	1.67	1.67	0.66	0.66

Table 42: Up- and Downstream processes before and after NewExt - Germany

[Euro-Cent / kWh]	Hard Coal		Oil		Gas	
	before NewExt	NewExt	before NewExt	NewExt	before NewExt	NewExt
SO_2	0.02	0.01	0.17	0.14	0.001	0.001
NO_x	0.01	0.01	0.04	0.05	0.01	0.02
Particulates	0.25	0.34	0.10	0.13	0.04	0.05
Sum air pollutants	0.27	0.36	0.31	0.33	0.05	0.07
CO_{2equiv}	0.21	0.21	0.15	0.15	0.10	0.10
Sum air pollutants and CO_{2equiv}	0.48	0.57	0.46	0.48	0.15	0.17

7.2.4 United Kingdom

Table 43: Power generation before and with NewExt - United Kingdom

[Euro-Cent / kWh]	Hard Coal		Oil		Gas	
	before NewExt	**NewExt**	**before NewExt**	**NewExt**	**before NewExt**	**NewExt**
Public health						
Mortality - YOLL	0.59	0.61	0.58	0.54	0.03	0.05
of which TSP	0.10	0.13	0.01	0.01	0.00	0.00
SO_2 as sulphates	0.33	0.25	0.38	0.29	0.00	0.00
SO_2 as SO_2	0.04	0.02	0.04	0.02	0.00	0.00
SO_2 as total	0.37	0.27	0.42	0.31	0.00	0.00
NO_x as total	0.12	0.21	0.15	0.21	0.03	0.05
NO_x (via ozone)	-0.07	-0.03	-0.03	-0.01	-0.01	-0.01
NO_x (via nitrates)	0.19	0.24	0.17	0.22	0.05	0.06
Morbidity	0.08	0.14	0.16	0.20	-0.02	-0.01
of which TSP, SO_2, NO_x	0.25	0.31	0.23	0.26	0.02	0.03
NO_x (via ozone)	-0.17	-0.17	-0.06	-0.06	-0.04	-0.04
Crops	-0.07	-0.07	-0.03	-0.03	-0.02	-0.02
of which SO_2	-0.002	-0.002	-0.004	-0.004	0.00	0.00
NO_x (via acid and N dep.)	-0.003	-0.003	-0.001	-0.001	-0.001	-0.001
NO_x (via ozone)	-0.07	-0.07	-0.03	-0.03	-0.01	-0.01
Materials	0.03	0.03	0.02	0.02	0.002	0.002
CO_{2equiv}	1.66	1.66	1.16	1.16	0.75	0.75

Table 44: Up- and Downstream processes before and after NewExt - United Kingdom

[Euro-Cent / kWh]	Hard Coal		Oil		Gas	
	before NewExt	NewExt	before NewExt	NewExt	before NewExt	NewExt
SO_2	nd	nd	0.10	0.08	nd	nd
NO_x	nd	nd	0.04	0.06	nd	nd
Particulates	nd	nd	0.01	0.02	nd	nd
Sum air pollutants	nd	nd	0.15	0.16	nd	nd
CO_{2equiv}	0.16	0.16	0.09	0.09	0.02	0.02
Sum air pollutants and CO_{2equiv}	nd	nd	0.24	0.25	nd	nd

7.3 Application of the methodology for assessing the total impacts of air pollution

In order to demonstrate how the new methodology developed in Newext can be applied to purposes like accounting issues and to extend the analysis to the new member states, calculations from the EC project GreenSense were updated (Droste-Franke and Friedrich 2003). Additionally to the applied regional models, for the local exposure assessment sector and population density specific estimates were used which were derived from new calculations and results within former EC projects (Droste-Franke and Friedrich 2003), (Link et al. 2001), (Schmid et al. 2001). The results are shown in Table 45, subdivided into damages occurring inside and outside the EU-25. Furthermore, it is distinguished between the different damaging substances.

Table 45: Mortality effects and total damage costs due to human health effects caused by emissions within the EU-25 in 1998

Substance	Total anthropogenic emissions within the EU-25		Public power, cogeneration and district heating plants within the EU-25	
	Mortality effects [years of life lost]	**Human health damage costs[1] [million $Euro_{2000}$]**	**Mortality effects [years of life lost]**	**Human health damage costs[1] [million $Euro_{2000}$]**
Inside the EU-25				
Nitrates	700,000	53,000	74,000	5,500
Sulfates	510,000	38,000	290,000	22,000
Primary Particles (PM_{10})	820,000	62,000	50,000	3,700
Ozone and SO_2	32,000	7,500	10,000	290
Total (rounded)	**2,070,000**	**160,000**	**420,000**	**31,000**
Outside the EU-25				
Nitrates	70,000	4,000	8,000	700
Sulfates	80,000	7,000	50,000	3,000
Primary Particles (PM_{10})	20,000	1,000	5,000	400
Ozone and SO_2	6,000	1,800	1,000	140
Total (rounded)	**170,000**	**10,000**	**70,000**	**5,000**

[1] includes mortality as well as morbidity effects

The total mortality effects caused by anthropogenic emissions of the EU-25 states were estimated to about 2.2 million years of life lost in 1998. Assuming 5 years lost per case per affected person (European Commission 1999b), p. 248, this number corresponds to about 450,000 premature deaths. The total assessed human health effects from the emissions in the EU-25 states in 1998 add up to 170 billion $Euro_{2000}$ damage costs with a contribution of about 36 billion $Euro_{2000}$ caused by air pollution due to public power, cogeneration and district heating plants which is only slightly less than the 44 billion $Euro_{2000}$ caused by emissions from road transport. About 160 billion $Euro_{2000}$ occur within the EU-25 states which corresponds to about 2 percent of the GDP in 1998.

8 Discussion

From the results shown in Table 22 to Table 27 the following conclusions can be drawn:

For the investigated technologies, the updated methodology of NewExt leads to similar results as before these updates. This is due to the fact that the reduction of external costs due to the lower VLYL gathered from the questionnaire survey (50,000€ compared with 96,500€ for chronic mortality and 75,000 compared with 165,700€ for acute mortality) is compensated by the increased values of the exposure-response functions for ‘chronic mortality’ and ‘chronic bronchitis’. In general, the comparison of the more detailed results, e.g. Table 37, shows that the impacts on human health morbidity have increased while the impacts on human health mortality have decreased. However, the total, i.e. the sum of the external costs caused by different impact categories and the exact ratio of the results before and after NewExt depend on the composition of the pollutants and the location of the power plant.

The external costs evaluated due to the methodology before NewExt are significant lower compared with the results of National Implementation (see results in Table 29 to Table 35). External costs are reduced mainly due to updated exposure-response functions for nitrates regarding human health, and the exposure-response functions for ‘chronic mortality’. The results in Table 10 show that in previous times the creation of nitrates was overestimated. Due to the better solution of the underlying grid (smaller grid cells) and the updated background emissions of NO_x, SO_2 und NH_3 the EcoSense model calculates less nitrates. Moreover, a model which accounts for tropospheric ozone due to NO_x and NMVOC is now implemented. Hence, more accurate results for impacts due to ozone are calculated.

Depending on the respective technology the external cost vary in the investigated countries up to a factor of three. The external costs for the gas fuel cycle are in general very low. The result for the up- and downstream processes rely on damage factors with an approximation for local impacts (Table 20 and Table 21). It is desirable for future projects to know the exact location of the emissions of the different fuel cycles stages. Damages can than be assessed at the location of the emissions. This enables to account more precisely for impacts in the local area. For the same reason, the emission height is important and should be available.

Comparing the external costs of different technologies in different countries one should carefully compare the input data. In some cases the value for a pollutant was zero, in another case there was no data available (e.g. Table 40, no PM10 for oil and gas). In both cases the result for this pollutant in zero. Moreover, up- and downstream emissions were not assessed for all cases. So the sub totals displayed in Table 22 to Table 27 are often not the entire external costs caused by the corresponding technology.

It has to be emphasized that the results shown in Table 22 to Table 27 may not be representative for the respective technology or the corresponding country. Rather, the results shown in Table 22 to Table 27 display the evaluated external cost of a plant defined during the National Implementation project. The results shown in this report are dedicated to compare the improvements in methodology from the state-of-the-art of the National Implementation to the state-of-the-art due to the findings in NewExt and their impacts on the results.

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VIII SUMMARY OF NEWEXT RESULTS

Based on a survey in three countries in the European Union, new values to assess the value of a statistical life (of 1.05 Mio. € as central value and 3.3 Mio. € as upper bound) and the value of a life year lost (75 000 €, upper bound 225 000 €) have been derived.

By analyzing the decisions of policy makers and in addition public referenda, shadow prices for global warming (ca. 5 to 22 € per ton of CO_2) and exceedance of critical loads for eutrophication and acidification (ca. 100 € per hectare of exceeded area and year with a range of 60 – 350 €/ha * year) have been developed.

The analysis of pathways of substances in air, water and soil made it possible to include the damage caused by the release of further substances into the framework for calculating external costs. Damage costs per kg of emitted pollutant of 80 €/kg for arsenic, 39 €/kg for cadmium, 29-34 €/kg for chromium, 1600 €/kg for lead, 4 €/kg for nickel and 0,2 €/kg for formaldehyde have been estimated.

The analysis of severe accidents in the non-nuclear fuel chains revealed, that the external costs associated with fatalities caused by these accidents are very small for power plants operated within EU-15: 0.0003-0.0007 €-cent/kWh for fossil fuels, 0.00002 €-cent/kWh for hydropower; in non-OECD countries external accidents costs could be up to 0,1 €-cent/kWh for hydropower.

The use of these findings for estimating external costs leads to certain changes in results. For coal-fired plants, figures based on the new methodology are between 13 and 49 % lower than those calculated with the methods applied in the ‘National Implementation’ project phase of ExternE. The ranking of different technologies however does not change when using the improved methodology.

IX FURTHER RESEARCH NEEDS FOR THE EXTERNE PROJECT SERIES

The research of this project has shown that almost all elements of the ExternE methodology need to be further improved and updated. Some of these will be addressed by the project NEEDS going on in the 6th Framework Program.

- **Global warming**

This subject is so vast and complex, with such rapid accumulation of new knowledge, that the need for further research is obvious.

- **Atmospheric dispersion and chemistry**

The models of atmospheric dispersion and chemistry used by ExternE can be improved and updated due to new insights to further increase the credibility of the results.

- **Health impacts**

The assessments need continual updating because of the intense worldwide research on air pollution epidemiology. The monetary valuation of health impacts also needs to be improved, especially for mortality and chronic bronchitis.

- **Damage to buildings and materials**

The inventories of buildings and materials need updating, and so do the dose-response functions. A major gap is the valuation of damage to buildings and monuments of cultural value.

- **Acidification and eutrophication**

The monetary valuation is still very uncertain; furthermore critical loads data are not freely available. Other methodologies should be explored.

- **Amenity impacts**

Whereas the valuation of noise is well developed, the reduction of visibility is a potentially very significant impact that has been neglected by ExternE so far. The cost of visual intrusion has not yet been addressed either.

- **Land use**

Land use, for example by surface mines or by roads, can have very severe ecosystem impacts that should be evaluated.

- **Supply security**

Some work is being done in ExternE-Pol, but it will not be sufficient.

Other issues

For several impact categories quantification in monetary terms is very difficult, if not meaningless, in particular the storage of waste, nuclear proliferation and risks of terrorism. Alternative approaches may have to be explored for the internalization of such impacts.

X OTHER INFORMATION AND DISSEMINATION ACTIVITIES

Beside the project website of NewExt (http://www.ier.uni-stuttgart.de/newext/) established at the beginning of this project for internal and external information and communication, a permanent website http://www.externe.info/ with more general information about the ExternE project series has been built up at the beginning of 2002. It has been and will further be extended for this purpose (within the concerted action DIEM) in order to contain all information about methodology and existing results. This web site http://www.externe.info/ also forms the backbone of the dissemination activities; all available publications will be provided as electronic versions for a better diffusion of relevant results. Some of them are already available at http://www.externe.info/reports.html.

All these activities have been the task of work package 7, the dissemination of the project. The objectives to make the new methodological elements available to the scientific community and to the end users of the EU external costs accounting framework have been met by the four workshops having taken place within the concerted action DIEM - see the elaborate description at the website http://www.externe.info/diem.html that also includes all contributions of the workshops for stakeholders and end users for download.