TOOLS AND STRATEGIES FOR IMPROVING POLICY RESPONSES TO THE RISK OF AIR POLLUTION

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ABSTRACT

This paper offers a brief review of the need for cost-benefit analysis (CBA) and the available policy instruments for air pollution. To prioritize different possible actions, one needs to know which source of pollution causes how much damage. This requires an impact pathway analysis, i.e. an analysis of the chain emission $\rightarrow$ dispersion $\rightarrow$ dose-response function $\rightarrow$ monetary valuation. The methodology for this is described and illustrated with the results of the ExternE (External Costs of Energy) project series of the European Commission. Two examples of an application to CBA are shown: one where a proposed reduction of emission limits is justified, one where it is not. It is advisable to subject any proposed regulation to a CBA, including an analysis of the uncertainties. Even if the uncertainties are large and a policy decision may have to take other considerations into account, a well documented CBA clarifies the issues and provides a basis for rational discussion. One of the main sources of uncertainty lies in the monetary valuation of premature mortality, the dominant contribution to the damage cost of air pollution. As an alternative an innovative policy tool is described, the Life Quality Index (LQI), a compound indicator comprising societal wealth and life expectancy. It is applied to the Canada-wide standards for particulate matter and ozone. Regardless of monetary valuation, a 50% reduction of PM_{10} concentrations in Europe and North America has been shown to yield a population-average life expectancy increase in the order of 4 to 5 months.

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INTRODUCTION

Air pollution has a variety of undesirable effects on human health, on buildings and materials, on agricultural crops and on ecosystems. There is now a general scientific consensus that air pollution causes health damage. Furthermore, the projects undertaken to quantify the costs of environmental damage have all come to the essential conclusion that the cost of health impacts by far outweighs damage from all other categories (ExternE, 1998; ORNL/RFF, 1994; Rowe et al., 1995; Abt, 2000) for the classical air pollutants (PM, NO_x, SO_2, O_3, VOCs, CO).

To reduce air pollution, several government regulations have been enacted over the past decades and these regulations have played a crucial role in curbing the emission of pollutants. Even though the regulations have become far more stringent than they were in the past, there is some debate about whether the current standards provide sufficient protection for human health. If stricter regulations are to be enacted, there is a policy imperative to determine whether the benefits exceed the costs. The rational foundation for introducing new and stringent regulation to control air pollution could be enhanced considerably if the benefits are quantified.

Past decisions about environmental policy have generally been made without quantifying the benefits. Initially, increasing demands for cleaner air were met by technical developments (such as flue gas de-sulfurization) without prohibitive costs. A simple criterion seemed adequate for making decisions. This criterion was based on the idea that a toxic substance has no effect below a certain threshold dose. If that is the case, it is sufficient to reduce the emission of a pollutant below the level where the highest dose anywhere is below the threshold. Standards for ambient air quality were developed, for example by the World Health Organization, and industry was required to reduce the emissions to reach these standards.

However, the situation is changing. Epidemiologists have not been able to find “no-effect thresholds” for air pollutants. The most recent guidelines of the World Health Organization indicate that there seems to be no such threshold for particulate matter (PM). The available evidence suggests that the dose-response function may be linear at low dose for PM, and quite possibly for other air pollutants as well. At the same time the incremental cost of reducing the emission of pollutants increases sharply as lower emission levels are reached. Thus the question "how much to spend?" acquires growing urgency. General principles such as sustainable development or the precautionary principle provide no answer (except in their most extreme and totally impractical interpretation of demanding zero pollution) because the difficulties lie in the specifics of each situation. One needs cost-benefit analysis (CBA). Of course, a cost-benefit analysis of air pollution should include all benefits, not just those due to health impacts.

Environmental CBA is often controversial. The objection to environmental CBA is often based on the view that one should not assign monetary values to goods such as a beautiful landscape, the existence of a rare animal or human life. However, this objection is less persuasive if we consider monetary valuation not in terms of the intrinsic value of the item in question but the collective willingness to pay (WTP) to avoid losing the item. For instance our WTP (including ability to pay) to avoid the risk of a premature death is limited, even if we feel that the value of life is infinite.

Above all, a thorough and well-documented CBA can provide a systematic assessment of the consequences of a decision before it is too late, and by clearly exposing the assumptions, it facilitates informed discussion of disagreements; for these reasons a CBA should always be attempted for important decisions even if the uncertainties are large. A CBA should however be used with care, firstly because of the large uncertainties, secondly because it may be desirable to take non-monetary considerations into account, for example the perception of risks and the distribution of costs and benefits among the population. Having estimated monetary values for all the benefits that can be quantified, any remaining non-monetary considerations can be evaluated by means of multicriteria analysis. Thus, the benefits of a CBA lie in what is made transparent in the process of doing it, as well as in the answers it provides.

The extra cost of a cleaner environment must be paid, ultimately by tax payer or consumer. Even if immediate trade-offs do not cross budget categories, ultimately the money we spend on reducing the emission of pollutants is not available for other causes such as the education of our children.

Links can be subtle and unexpected. When evaluating a decision, it is necessary to consider the consequences of alternatives and unintended effects. For example, lowering the limit for the allowable emission of dioxins from waste incinerators will avoid some cancer deaths, but people will have to pay more for waste disposal. Such costs induce effects elsewhere in the economy. For example, in the USA Keeney (1994) has shown that for each $ 5 to 10 million of cost imposed by a regulation there will be on average one additional premature death due to this cost.

The value of achieving clean air objectives must be commensurate with the benefits and it is important to aim for a level that is optimal for society.

POLICY INSTRUMENTS

In recent years the term “external cost” has been widely used to designate the costs of environmental damage. The term damage cost is more appropriate since it avoids ambiguities that arise with the use of at least two definitions of external cost:

1) costs imposed on non-participants, that are not <u>taken into account</u> by the participants in a transaction;
2) costs imposed on non-participants, that are not <u>paid</u> by the participants in a transaction.

According to the first definition a damage cost is internalized if the polluter reduces the emissions to the socially optimal level, for example as a result of a regulation that imposes an emission limit. The second definition requires, in addition, that the polluter compensate the victims for any damage, for example by paying a pollution tax. In either case, the level of emissions is equal to the social optimum. But the corresponding damage cost is external only according to the second, not the first definition. Some economists have used the term “relevant externality” to designate the portion (if any) of the damage cost that is greater than the social optimum.

Between these two levels of internalization there is a substantial difference in the costs borne by the polluter, as shown by Desaigues and Rabl (2001). In practice the argument for full internalization according to the second definition loses its justification, since it is almost impossible to compensate the victims correctly (identifying who suffers how much damage is too difficult and uncertain).

Of course, government intervention is necessary to reduce the emission of air pollutants. There are various possible policy instruments and many different ways of implementing them. For brevity we list in Table 1 the principal types of regulations or policy instruments that have been used for this purpose. Some of the options when adopted have a direct impact on the emissions whereas others such as eco-labels and portfolio standards can affect emissions indirectly by reducing the consumption of materials or energy. At one end of the spectrum, characterized as “command-control”, are regulations that impose rigid and specific constraints (e.g. a limit on the concentration of SO_2 in the flue gas of power plants). At the other end are market mechanisms: the government can propose certain general goals or targets, for instance a national emission ceiling or a tax per ton of SO_2, and let the markets respond by providing an appropriate solution. The middle column of the table indicates whether the regulations are based on command-control (C) or on market mechanisms (M).

A command-control approach (provided, of course, that compliance is enforced) yields predictable results (e.g. the specified reduction of SO_2), but often the costs are high because all polluters must take the same action. The costs of pollution abatement depend on specific local circumstances and vary greatly from one polluter to another. For instance, they are much higher for an industry that must install an expensive retrofit than for one that can include the pollution control equipment in the early design phase of a new factory. Under a pollution tax, there is some flexibility and the owner can choose and optimize how much of the pollutant to remove by abatement equipment, paying the tax on whatever remains. Whereas a pollution tax achieves reductions at the lowest possible overall cost for society per avoided unit of pollutant (highest economic efficiency), the magnitude of the realized reduction is difficult to predict.

Table 1. Policy instruments for reducing air pollution.

Type	C or M	Examples
Limits on emission of pollutants	C	max. mg SO_2 per m^3 of flue gas; max. g CO per km driven by cars
Choice of technologies	C	usually by demanding "Best Available Technology" (BAT), e.g. flue gas desulfurization for coal or oil fired boilers
Broad initiatives to reduce emissions of an entire sector or country	depends on implementation	The National Emission Ceilings of the EU
Subsidies for clean technologies	M	tax credit for wind and solar in California during 80s
Eco-labels	M	"printed on recycled paper"; "no chlorine used"; "energy star" label for computers
Pollution taxes	M	€/ton of a pollutant
Tradable permits	M	government sets cap on number of permits (e.g. ton of SO_2), polluters can trade these permits
Portfolio standards	M	government sets minimum % for the market share of a clean technology, e.g. "zero emission" vehicles in California, or “green kWh” from solar energy, and industry adjusts the prices to achieve these goals.

Tradable permits are a policy instrument that combine the highest economic efficiency with predictable results. Under this system the government issues permits for a specified quantity of a pollutant that may be emitted in a region, and industry can freely buy or sell these permits. There are several variants, the two main distinctions being whether the government sells the permits at an auction or gives them away (for instance to each polluter according to last year’s emission). Obviously, industry prefers the latter. In the USA tradable permits have been used for SO_2 for the last decade already, with great success: the cost per avoided kg turned out to be much lower than under the previous regime of command-control.

There has been some opposition to the idea, especially from people who misunderstand “permit” to mean unlimited license to pollute and do not recognize that the most widely used regulation, namely emission limits, is de facto a permit (to emit up to the specified limit) that is given away freely but cannot be traded. And tradable permits are not an unlimited license to pollute: the polluter incurs a cost for each kg of emitted pollutant.

At this point we can make several general recommendations:

- Market instruments yield a better allocation of societal resources than command-control.
- Tradable permits that are given away free are preferable to permits that are auctioned or to pollution taxes because they imply much smaller changes in the costs incurred by industry (for the same reduction of emissions) and thus less perturbation of the economy when they are introduced or modified (Desaigues and Rabl, 2001).
- Subsidies are treacherous because experience has shown how difficult they are to remove when they are no longer justified; they should be used only if automatic termination can be guaranteed.
- Specific policy proposals should be carefully evaluated before application to ensure that they are cost-effective and that they will not entail unexpected harmful side effects. In the past most regulations had not been subjected to a cost-benefit analysis before passage, and some have not been well chosen (see the examples later in this paper).
- Compliance with government regulations must be verified and enforced. It is desirable for the process to be clear and transparent. In this context one could add that major polluters should be required to post their emissions data on the internet rather than treating them like an industrial secret as has so often been the case, for example with utility companies in many countries of Europe.

THE NEED FOR IMPACT PATHWAY ANALYSIS

Policy decisions must act on the sources of pollution. To provide adequate guidance to the formulation of policies it is not sufficient to calculate the damage per exposure; one needs to know which source of a pollutant causes how much damage. This requires an impact pathway analysis (IPA), tracing the passage of the pollutant from where it is emitted to the affected receptors (population, crops, forests, buildings, etc.), as shown in Figure 1. The impacts and costs are summed over all receptors of concern. Since the dispersion of air pollutants is significant over distances of hundreds to thousands of km, the analysis must account for all impacts at the scale of an entire continent.

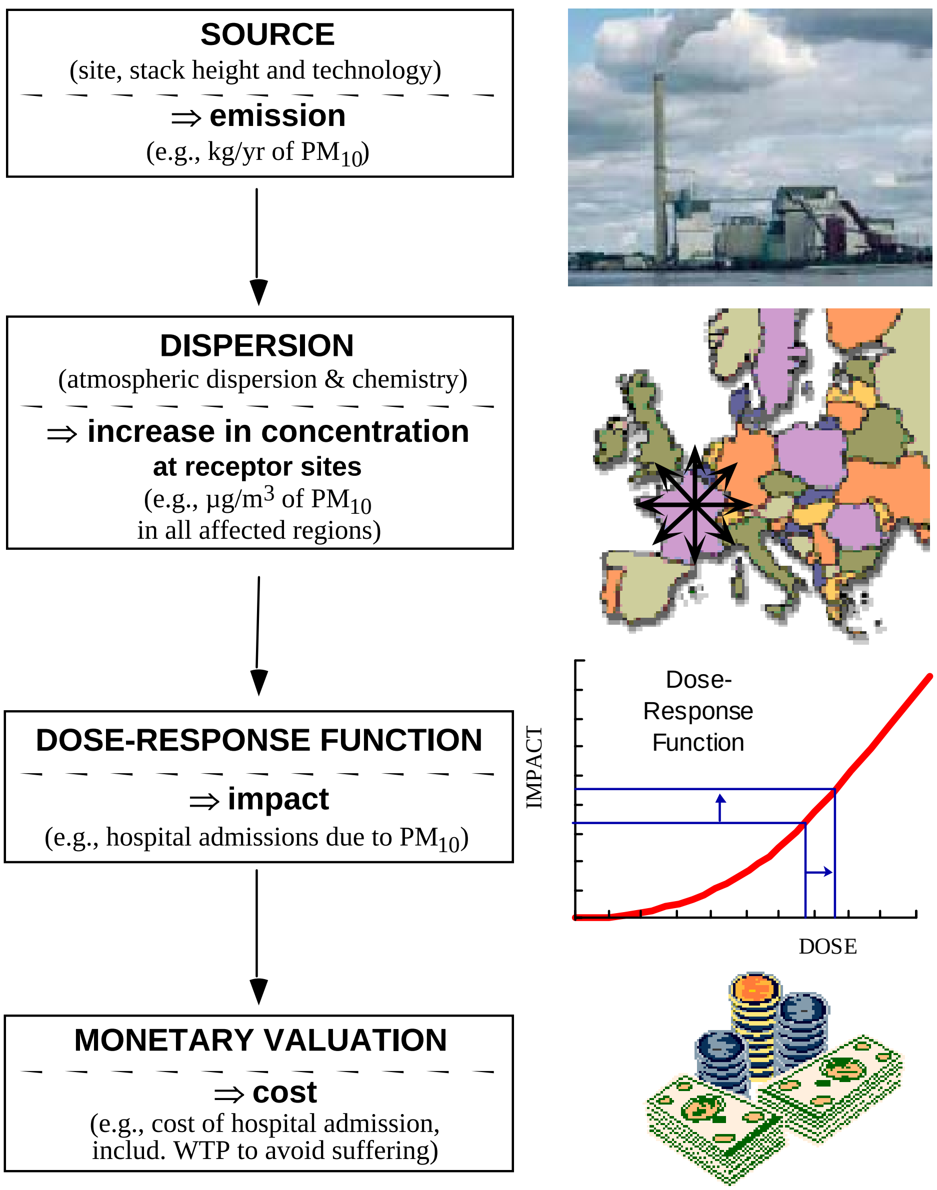

Figure1. Impact pathway analysis for the example of an air pollutant.

This requires a multidisciplinary system analysis with inputs from engineers, dispersion modelers, epidemiologists, and economists. The largest and most up-to-date effort of this kind is the ExternE (External Costs of Energy) project series of the European Commission (EC) which began in 1991 and is still continuing. Initially ExternE was carried out jointly with the USA (ORNL/RFF, 1994), but the US participation stopped in 1995. Some work analogous to ExternE has continued (USDOE, 2003), and other projects that parallel efforts of ExternE include the work of IIASA on the RAINS model (see e.g. Mechler et al., 2002), of Kuenzli and colleagues (Kuenzli et al., 2000), and of Ostro, Rowe and co-workers. All of these studies find that health impacts account for well over 90% of the quantifiable damage cost of air pollution other than global warming, and that among health cost the cost of mortality is by far the largest. Therefore the monetary valuation of air pollution mortality is a central issue. First we present some results of ExternE followed by a discussion of monetary valuation. A very brief summary of the methodology is given in Appendix A; for more detail the reader is referred to the ExternE reports or to review papers (e.g. Rabl and Spadaro, 1999).

RESULTS OF EXTERN E

Damage Costs per kg of Pollutant

The output of an impact pathway analysis is the damage cost per kg of an emitted pollutant, given the site and conditions of the source. Some results for typical French conditions are presented in Figure 2. For primary pollutants there is a strong dependence on site of source and height (h) of source above ground; for secondary pollutants the variation with site is much smaller (about +-30% for sources in France) and the variation with h is negligible. Note that the results cited in different reports of ExternE can be somewhat different because the methodology has been evolving. The key assumptions for the numbers shown here are listed in Table 2.

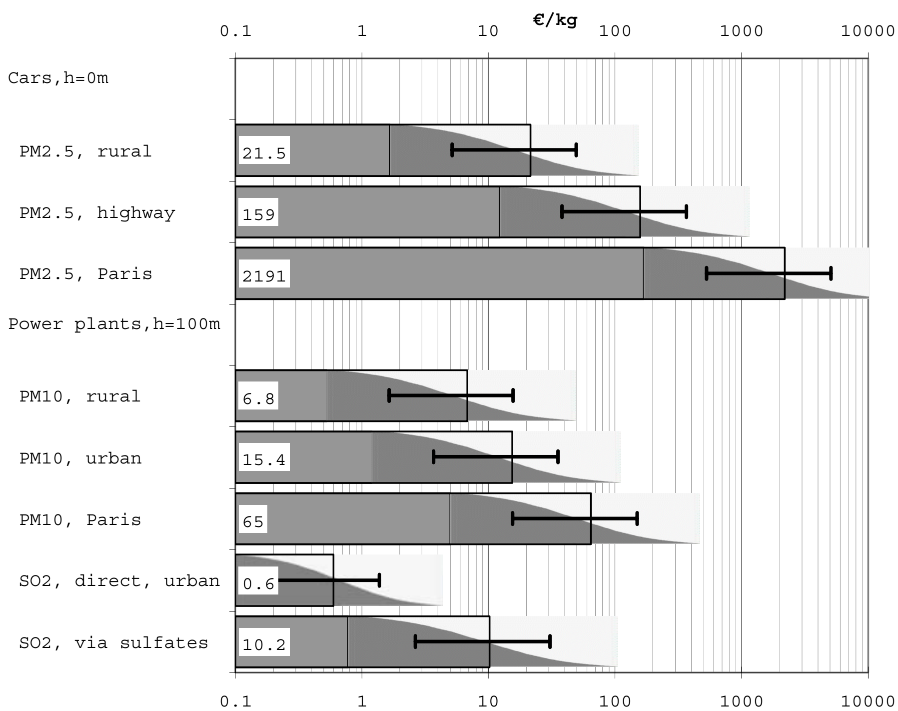

Figure 2. Typical damage costs for PM and SO_2 (values of ExternE 1998 for France), together with the uncertainty range. The gray curve indicates the probability that the true cost is below a specified value. The mean damage costs are shown in the labels. On a log plot the distribution is symmetric about the median which is lower than the mean. The error bars indicate 1 geometric standard deviation (68% confidence interval).

Table 2. Key assumptions for the damage costs cited in this paper.

Atmospheric dispersion models	
Local range:	gaussian plume models ISC and ROADPOL.
Regional range (Europe):	Harwell Trajectory Model as implemented in ECOSENSE software of ExternE. Ozone impacts based on EMEP model
Impacts on health	
Form of dose-response functions	Linearity of incremental impact due an incremental dose (e.g. $\mu g/m^3$ ambient pollutant concentration) for all health impacts.
Chronic mortality	Dose-response function slope = 4.1E-4 YOLL (years of life lost) per person per year per $\mu g/m^3$ derived from increase in age-specific mortality due to $PM_{2.5}$ (Pope et al. 1995), by integrating over age distribution.
Acute mortality	For SO_2 and ozone, with 0.75 YOLL per death.
Nitrate and sulfate aerosols	Dose response functions for nitrates same as for PM_{10}. Dose response functions for sulfates same as for $PM_{2.5}$ (slope = 1.7 times slope of PM_{10} functions).
Micropollutants	Only cancers have been quantified (As, Cd, Cr, Ni, dioxins, benzene, butadiene).
Monetary valuation	
Valuation of premature death	Proportional to reduction of life expectancy, with value of a YOLL (years of life lost) derived from VSL = 3.4 M€: 96.5 K€/YOLL for mortality.
Valuation of cancers	0.45 M€ nonfatal cancers, 1.5 to 2.5 M€ (depending on YOLL) fatal cancers, 1.5 M€ average for cancers from chemical carcinogens.
Discount rate	3% unless otherwise stated; results for nuclear are shown for 0% "effective discount rate" (=discount rate – escalation rate of cost).

YOLL = years of life lost, VSL = value of statistical life

Uncertainties

The uncertainties in this domain are very large. They should be evaluated, to give decision makers an idea of the reliability of the results. Uncertainties can be grouped into different categories, even though there may be some overlap:

- data uncertainty (e.g. slope of a dose-response function, cost of a day of restricted activity, and deposition velocity of a pollutant);
- model uncertainty (e.g. assumptions about causal links between a pollutant and a health impact, assumptions about form of a dose-response function (e.g. with or without threshold), and choice of models for atmospheric dispersion and chemistry);
- uncertainty about policy and ethical choices (e.g. discount rate for intergenerational costs, and "value of statistical life");
- uncertainty about the future (e.g. the potential for reducing crop losses by the development of more resistant species);
- idiosyncrasies of the analysts (e.g. interpretation of ambiguous or incomplete information).

The first two categories (data and model uncertainties) are of a scientific nature. They are amenable to analysis by statistical methods, combining the component uncertainties over the steps of the impact pathway, to obtain formal confidence intervals around a central estimate. For this, ExternE followed an approach based on lognormal distributions and multiplicative confidence intervals. For quantifying the sources of uncertainty, a survey was carried out of experts and relevant information available in the literature. The results of this analysis are shown Figure 2; the error bars are one-geometric standard deviation intervals around the median estimate. The largest sources of uncertainty lie in the dose-response functions for health impacts and in the value of a life year. Details can be found in Rabl and Spadaro (1999).

One of the sources of uncertainty lies in the difficulty of identifying exactly which air pollutant causes how much damage, since epidemiological studies are carried out under real conditions of exposure to a mix of pollutants. Thus the total health damage attributable to all air pollutants is probably more certain than the individual damage costs for each pollutant. This becomes important when evaluating the effects of sources that generate mixtures which differ markedly from the general urban mixture as a whole. For example, the estimated damage costs of gas-fired power stations are much more dependent on judgments made about the health effects of NO_2 as a gas, and of nitrates as secondary particles, than are the estimated costs of coal-fired stations, where judgments about SO_2 and sulfates are more influential.

One should note that the full uncertainty is larger than the data and model uncertainties that have been quantified explicitly and shown in Figure 2.

Use of ExternE Results

Gradually the results of ExternE are diffusing into the world of decision makers. For example, ExternE is recognized as the reference for comparative risk assessment by agencies such as the International Atomic Energy Agency. In the EU, ExternE is increasingly used as input to environmental decisions, for example via cost-benefit analyses (Holland, 2001). Of course, a cost-benefit analysis of air pollution should include all benefits, not just those due to health impacts.

Figure 3 is a comparison of social costs and benefits for a proposed reduction of the emission limit for particulate matter (PM) emitted by cement kilns that use waste as fuel, one of the issues under discussion in formulating the new EC Directive on the incineration of waste. Even the upper bound of the benefit is lower than the lower estimate of the abatement cost. Clearly this proposed emission limit cannot be justified by a cost-benefit criterion (see Rabl (2000), a paper which contributed to the decision to require only 30 mg/m^3, not the lower limit of 15 mg/m^3 which had been proposed).

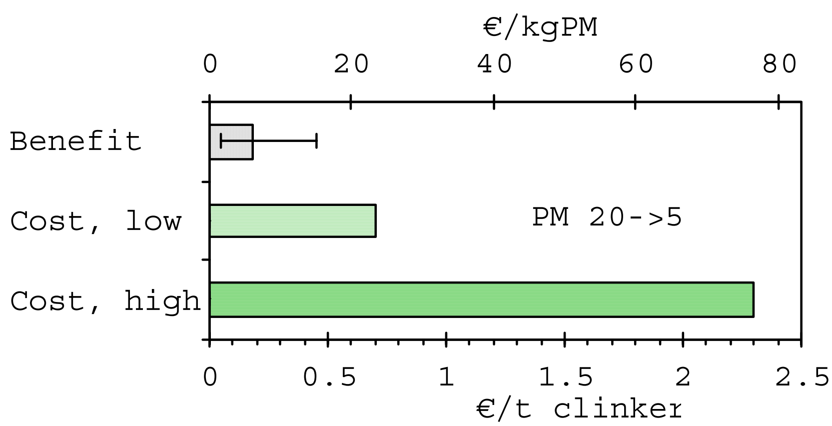

Figure 3. Comparison benefits for a reduction of average emission from 20 to 5 mg/m^3 (of emission limit from 50 to 15 mg/m^3). Costs and benefits are shown on two scales: per $t_{clinker}$ (bottom) and per kg_{PM} (top). Error bar indicates uncertainty of benefit.

By contrast Figure 4 shows an example where the proposed reduction of emission limits is justified (Rabl et al., 1998a). This figure compares cost and benefit for a reduction of emission limits for PM and SO_2 from municipal solid waste incinerators: the limit for PM being reduced from 30 to 10 mg/m^3 and the one for SO_2 from 300 to 50 mg/m^3. Whereas in Figure 3 the benefit is shown for a single site, cement kilns being typically located in rural sites, in Figure 4 it is appropriate to show at least three sites for incinerators: Paris (population of about 10 million, including suburbs), a typical urban site and a rural site. For all of these sites the benefit outweighs the cost. These reduced emission limits have been incorporated in the above mentioned EC Directive on the incineration of waste.

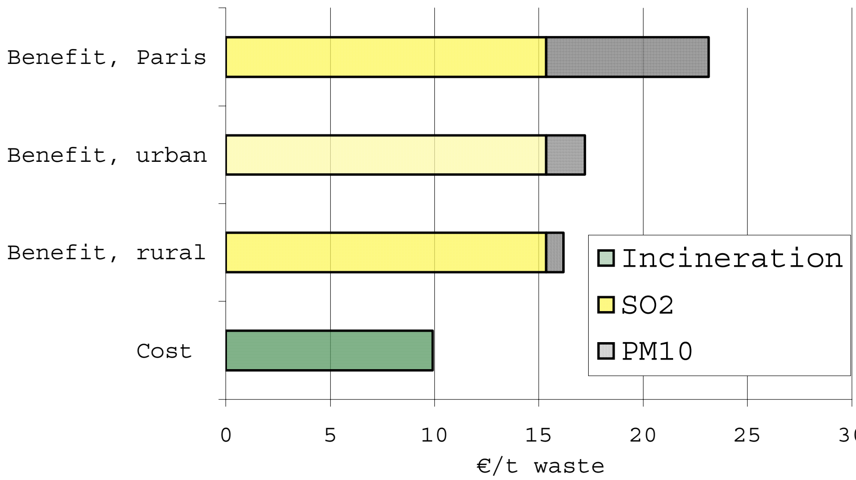

Figure 4. Cost and benefit for a reduction of emission limits for PM and SO_2 from municipal solid waste incinerators.

The difference between Figures 3 and 4 arises from the differences in technology. For waste incinerators the abatement technology under consideration reduces both PM and SO_2. The technology for cement kilns is different and reduces only PM (apart from rare exceptions cement kilns emit no SO_2).

As another application of ExternE we mention a CBA of the particle filter for diesel buses (Rabl et al., 1998b), where we found that this technology is justified by a wide margin in Paris, but not for rural bus routes. For a more recent CBA of particle filters, also for trucks and passenger cars, see Massé (2003); it finds that this technology is now mature and with current costs can be justified even for passenger cars.

The lesson to be drawn from these examples is that in some cases stricter limits for the emission of pollutants are clearly justified, in others they are not. A careful analysis should be carried out before implementing new policies, to avoid wasteful allocation of resources.

As emphasized in the preceding section, the uncertainties of the benefits are very large. But the abatement costs are also uncertain. For mature and widely used control technologies, the abatement costs are reasonably well known, but for the more important case of new technologies or new applications their estimation is often very difficult and uncertain. Often the industries concerned by a proposed reduction of pollution may not have all the required information, and if they do they may prefer to keep it confidential.

MONETARY VALUATION OF AIR POLLUTION MORTALITY

Ground rule

The goal of the monetary valuation of damages is to account for all costs, market and non-market. For example, the valuation of an asthma attack should include not only the cost of the medical treatment but also the willingness-to-pay (WTP) to avoid the residual suffering. If the WTP for a non-market good has been determined correctly, it is like a price, consistent with prices paid for market goods. Economists have developed several tools for determining non-market costs; of these tools contingent valuation (CV) has enjoyed increasing popularity in recent years (Mitchell & Carson, 1989). The results of well conducted studies are considered sufficiently reliable to be used in policy applications – see previous Section.

"Value of statistical life" VSL and Value of a Life Year

The “value of statistical life” (VSL) is such an important parameter for public policy that economists have carried out numerous studies to determine it (far more than a hundred, the vast majority in anglo-saxon countries). There are basically three approaches that have been used to determine how much an individual is willing to pay to reduce a risk of premature death:

i) comparisons of the relation between wage and risk in different professions with different levels of risk;
ii) the purchase of goods such as smoke detectors that reduce risks;
iii) contingent valuation (direct interrogation by means of questionnaire or interview).

The estimation of VSL has been a challenging and controversial topic in risk analysis. Empirical estimates of VSL typically range from $1 million to $10 million as reported in a review of literature (Tengs et al., 1995). For example, in the cost-benefit analysis of a Canadian Air Quality Standard, $4.1 million is an age-adjusted central estimate of VSL (CWS, 1999).

There has been no single official value used and recognized by all government agencies. ExternE takes VSL as the average of all the VSL studies that had been carried out in Europe when the ExternE project started in 1991. After adjustments for inflation, this amounts to 3.1 M€ in 1995. It is interesting to note that in the USA EPA uses values that are about twice as high. In France, by contrast, the report of Boiteux (2001) on external costs of transport recommends 1.0 M€.

A crucial question for air pollution mortality is whether one should simply multiply the number of premature deaths by VSL, or whether one should take into account the years of life lost (YOLL) per death. All studies before 1996, and in the USA until now, have done the former. The valuation issue is linked to the epidemiological one, of whether the estimates of mortality impacts are derived from studies of very recent pollution only (‘acute mortality’, from time series studies) or from studies of longer-term exposure (‘chronic mortality’, from cohort studies). Studies before 1996 quantified acute mortality only. ExternE (1995) estimated chronic mortality also, but only as a sensitivity analysis. By 1998 ExternE had incorporated chronic mortality estimates into its core analyses. They proved to be the dominant health effect. This is now the generally accepted approach.

There are several reasons why number of deaths × VSL is not appropriate for air pollution mortality.

a. First of all, premature deaths from air pollution tend to involve far fewer YOLL per death than accidents (on which VSL is based). This is certainly the case for acute mortality, where the earlier (‘extra’) deaths occur principally among people who are older and, almost certainly, particularly vulnerable for their age. The time series studies do not provide any direct information about YOLL, but it is widely understood that this is short, typically several months rather than many years.
b. Secondly, air pollution is a contributing, not the only, cause of the mortality of individuals, and the sum of deaths due to all contributing causes would be far greater than the total number of deaths in a population; by contrast, YOLL from different causes can be added to yield a total that is meaningful.
c. Furthermore, estimates of how long-term exposure to pollution affects mortality are best carried out using life table or life expectancy methods. These lead naturally to impacts in terms of total YOLL in a population, rather

than to ‘extra’ deaths (ExternE 1998; Hurley et al., 2000; Leksell and Rabl, 2001; COMEAP, 2001). Indeed, the total number of deaths due to air pollution cannot be determined whereas the total number of YOLL can (see, e.g., (COMEAP, 2001; Rabl, 2003) – the available methods cannot distinguish whether only some individuals are affected, with a substantial loss of LE, or whether everybody is affected; in the latter case the number of deaths due to air pollution would be equal to the total number of deaths.

Until now there have been almost no published studies that determine the value of a YOLL directly, by contrast to the numerous VSL studies, most of them based on accidents. Therefore ExternE has derived the value of a YOLL from VSL, by assuming that VSL is the present value of a discounted series of annual YOLL values. The ratio of the resulting value of a YOLL and VSL depends on the discount rate; it is typically in the range of 20 to 30. ExternE (1998) uses 84000 $€_{1995}$ per YOLL. One of the tasks of the current phase of ExternE is a contingent valuation (CV) to determine the value of a YOLL directly. Results will be published soon. Another study with similar aims, for the UK Government (DEFRA), by Mike Jones-Lee and colleagues, is also nearing completion and results will be available shortly.

ExternE also assumes 0.45 M€ for nonfatal cancers, and 1.5 to 2.5 M€ for fatal cancers (depending on the YOLL for each cancer type).

It is reasonable to think that the value of a YOLL should contain an adjustment for age to account for the fact that reduction in air pollution lowers death rates primarily among older persons. However, the empirical basis for age adjustment is weak and the practice is controversial: most older people do not like to hear that their life years are valued less. Furthermore, a discount factor should be applied because of latency, i.e. the time between exposure and premature death. Such factor should account both for the time value of money and for the perceived utility of an extra life year in the future. At the present time the empirical evidence on which to base judgments about the latter is weak; the underlying cohort studies are not informative about this aspect.

Current estimates of the monetary value of a life year lost due to air pollution are extremely uncertain. Furthermore, even when or if reliable CV studies for this purpose become available, they will be limited to developed countries. There is an increasing demand for the valuation of mortality in the rest of the world where CV will not be practical in the foreseeable future. Therefore it is advisable to develop alternative methods.

One option is to use values implicit in public decisions, for example in the health sector; this is described in Appendix B. However, one finds that the implied value of “cost per life year” can vary by about six orders of magnitude. Such estimates tend to reflect subjective decisions (preferences) of program administrators, often under the influence of unrepresentative or ill-informed pressure groups, rather than being representative of tradeoffs or peoples’ willingness-to-pay for risk reduction. More importantly, the nature of risks in many life saving interventions is different than that associated with environmental interventions.

Another, very promising alternative is the life quality index (LQI), described briefly in the next section, with more detail about its derivation in Appendix C.

MONETARY VALUATION USING THE LIFE QUALITY INDEX (LQI)

Key Principles of the Life-Quality Index

The Life Quality Index (LQI) is a compound social indicator comprising societal wealth and longevity. It can also be interpreted as a utility function that is consistent with several principles of decision analysis. It has recently been applied to the cost-benefit analysis of pollution control programs (Pandey and Nathwani, 2003).

The proposed framework is intended to satisfy some basic reasoning and principles of risk management in public interest, namely, accountability, maximum net benefit, compensation and life measure, which have been discussed in detail elsewhere (Nathwani et al., 1997; Nathwani and Narveson, 1995). It incorporates the following principles:

(i) A unified rationale for application to all risks, if we are to have a working basis for practical professional action in society's interest when risks to life, health or property are important.
(ii) Maximizing the total expected net benefit to society. This principle has been accepted as fundamental to cost-benefit analysis. It satisfies the utilitarian concept of welfare, i.e., the greatest good for the greatest number. A simple and meaningful test of the effectiveness of allocation of scarce resources is: how much life saving does risk reduction buy, and could the same resource, if directed elsewhere, bring a better gain for society as a whole?
(iii) Compensation to ensure implementation of a policy is socially beneficial where there is a need to compensate the losers.
(iv) Enhancement of a relevant measure of life by maximizing the net benefit in terms of quality of life in good health for all members at all ages.

Definition of the Life-Quality Index

The LQI for a society is derived (see Appendix C) as

$$L = G^q E \tag{1}$$

where G is the real gross domestic product per person/year, E is the age-adjusted life expectancy in the country, and q is the elasticity of utility of consumption. q is related to a measure of labor productivity; for industrialized nations a typical value is 0.15.

The LQI consists of two major indicators: the real gross domestic product per person as a measure of resources and the quality of life (UN, 1990), and life expectancy which is a validated universal indicator of social development, environmental quality and public health (Gulis, 2000). Both indicators have been in use for half a century to express the wealth and health of a nation in numbers, and they are reliably measured.

Judging Risk with the Life Quality Index

Any project, program or regulation that materially affects the public by modifying risk through expenditure will have an impact on the Life-Quality Index. The net benefit criterion requires that a small change in the LQI due to a project or regulation should be positive or,

$$\frac{dL}{L} = q\frac{dG}{G} + \frac{dE}{E} \geq 0 \tag{2}$$

Here *dG* may represent the monetary cost of implementing a regulation (*dG* negative) or the monetary benefits that arise from a project (*dG* positive). The term *dE* is the change in life expectancy due to a change in the level of risk to the population associated with a project or, regulation.

The concept of societal WTP originates from the definition of compensating variation by Hicks (1939). It is the sum received by or from the individuals which, following a welfare change, leaves them at their original level of welfare. It can obtained from Eq.(4) by setting $dL/L = 0$ and rearranging the terms as

$$(-dG) = \frac{G}{q}\frac{dE}{E} \text{ (\$/person/year)} \tag{3}$$

Suppose benefits of a safety regulation are received by a population of size *N*, the aggregated value of societal WTP, i.e., the amount that will not alter the population life-quality (*C*) is equivalent to

$$C = (-\,dG) \times N = \frac{NG}{q}\frac{dE}{E} \text{ (\$/year)} \tag{4}$$

We propose the LQI-based measure of societal WTP for the valuation of mortality reduction in the cost benefit analysis of air-quality standards.

Application to the Canadian Air Quality Standard

Pandey and Nathwani (2003) have applied the LQI model to calculate monetary equivalent of benefit of reduction in mortality resulting from scenarios of improving air quality, which were studied during the development of a Canada Wide Standard (CWS) (CWS, 1999). The results of a cost-benefit analysis based on a simple multiplication of VSL and number of deaths are summarized in Table 3. Starting from the numbers in this study, with LQI the results in Table 4 were obtained for three rates of time preference r_{tp} (discounting of life years), 0%, 2% and 4%. The benefits associated with options to reduce particulate matter always outweigh the pollution control costs, as evident from benefit/cost ratios ranging from about 2 to 26. On the other hand, for all ozone options, these ratios are less than one, and so they do not satisfy the LQI criterion. Assuming that a rate of time preference of 2% is representative, the overall benefit/cost ratio of the Standard turns out as 1.9. Consideration of the effect of time preference rate is important, as benefit estimates can vary quite strongly when this rate changes by as little as one percent.

Table 3. Valuation of benefits of the Canada Wide Standard based on a simple multiplication of VSL and number of deaths, in Canadian $\$_{1996}$ (CWS, 1999).

Target Pollutant Level	**Avoided Mortality**	**Benefit of avoided mortality** [a]	**Cost**	**Benefit/Cost Ratio**
	(death/yr)	(million C$/yr)	(million C$/yr)	
$PM_{10}/PM_{2.5}$ ($\mu g/m^3$)				
70/35	1,021	4,186	170	24.6
60/30	1,639	6,720	620	10.8
50/25	2,790	11,439	1,600	7.1
Ozone (ppb)				
70	167	685	790	0.9
65	203	832	1,871	0.4
60	239	980	6,502	0.2
CWS				
$PM_{10}/PM_{2.5}/O_3$				
60/30/65	1,842	7,552	2,491	3.0

[a] Using central estimate of VSL = C$ 4.1 million/person; base year 1996, and discount rate 5%

Table 4. Cost-benefit analysis of Canada Wide Standard for Air Quality (CWS, 1999) using LQI approach.

Target Pollutant Level	**Cost (million C$/yr)**	**Benefit/Cost Ratio**		
		r_{tp} = 0%	r_{tp} = 2%	r_{tp} = 4%
$PM_{10}/PM_{2.5}$ ($\mu g/m^3$)				
70/35	170	26.6	15.6	9.0
60/30	620	11.7	6.9	4.0
50/25	1,600	7.7	4.5	2.6
Ozone (ppb)				
70	790	0.9	0.5	0.3
65	1,871	0.5	0.3	0.2
60	6,502	0.2	0.1	0.1
CWS[3]				
$PM_{10}/PM_{2.5}/O_3$				
60/30/65	2,491	3.3	1.9	1.1

CONCLUSIONS

1. In response to increasing evidence that particulate air pollution and ground level ozone have adverse impacts on public health and environment, stringent air-quality standards are under development worldwide.
2. The economic justification for pollution control programs largely rests on two aspects: using cohort studies to quantify the effects of long-term exposure on mortality, an approach which is now widely accepted; and the monetary valuation of reduction in mortality, which is a critical and controversial element of cost-benefit analysis.
3. The most relevant damages caused by air pollution (other than global warming) can be quantified and monetized using the impact pathway analysis of ExternE, albeit the uncertainties are large.
4. In many cases the accuracy of ExternE estimates is sufficient to provide guidance to a decision-maker, despite the uncertainties. For often, the policy issue or a problem related to the resolution of an environmental concern involves a “yes or no” type of question: “is the benefit greater than the cost?”
5. One of the main sources of uncertainty lies in the monetary valuation of air pollution mortality. The widely used “value of statistical life” is based on accidents and not appropriate, and the available estimates of the value of a life year lost due to air pollution are still very uncertain.
6. To circumvent the uncertainties of the valuation of air pollution mortality, the Life Quality Index LQI is proposed as an innovative policy tool because it allows integration of the key issues (discounting of life years, competing mortality risks, inter-temporal tradeoffs, age-dependent risks, and willingness to pay) in a consistent and transparent manner to support a credible analysis.
7. Assessment of potential health benefits of environmental policies, based on ExternE results, has clearly shown the gain in life expectancy that could be achieved in Europe and North America if the concentration of PM_{10} is reduced by 50%: the gain would be roughly 4 to 5 months (averaged over the entire population). This is a finding of enormous significance when compared to other measures for improving public health.
8. It is highly desirable to subject proposed environmental strategies to a cost-benefit analysis to help avoid costly mistakes and ensure that our scare resources are spent wisely. The necessary tools are now available. Of course, cost-benefit analysis in this domain is fraught with risks due to the large uncertainties and the subjective nature of some of the assumptions that may be made by the analyst. Whereas cost-benefit analysis must be used with caution, and not as the only criterion for a decision, it does provide a valuable framework to help clarify the issues.

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Appendix A. The Methodology of ExternE

Impact Pathway Analysis (IPA)

For the analysis of fuel chains, ExternE has coupled IPA with life cycle assessment (LCA), a tool that has been widely used for environmental analysis. The key idea of LCA is to take into account all the stages in the life cycle of a process or product. This is illustrated in Figure A1 for the example of electricity production. Whether an IPA of a single source or an LCA of an entire cycle is required, depends on the policy decision in question. For finding the optimal limit for the emission of SO_2 from a coal fired power plant, an IPA is sufficient, but the choice between coal and nuclear involves an LCA.

→ real impacts for each stage (site specific)
Goal: evaluate the entire matrix

Steps of impact pathway analysis → **Stage of fuel chain** ↓	Emission	Dispersion	Exposure-response function	Economic valuation
Fuel extraction				
Fuel transport				
Power plant				
Transmission of electricity				
Management of wastes				

Life cycle assessment: first sum over emissions ↓ Σ then → × multiplication by **"potential impact"** indices

Figure A1. Relation between impact pathway analysis and current practice of most LCA, illustrated for the example of electricity production From Rabl and Spadaro (1999).

In principle, the damages and costs for each pollution source in the life cycle should be evaluated by a site-specific IPA. But in practice almost all LCA has taken the shortcut of first summing the emissions over all stages and then multiplying the result by site-independent impact indices. Also, most practitioners of LCA reject the concept of monetary valuation, preferring instead to use non-monetary indicators of "potential impact" that are based on expert judgment.

Dispersion modeling

Since for most air pollutants other than the globally dispersing greenhouse gases, atmospheric dispersion is significant over hundreds to thousands of km both local and regional effects must be taken into account. ExternE has therefore used a combination of local and regional dispersion models. For dispersion over the local range (< 50 km from the source) two gaussian plume models have been used: ISC (Brode and Wang, 1992) for point sources such as power plants, and ROADPOL for lines sources (emissions from transport) (Vossiniotis et al., 1996).

At the regional scale one needs to take into account the chemical reactions that lead to the transformation of primary pollutants (i.e. the pollutants as they are emitted) to secondary pollutants, for example the creation of sulfates from SO_2. Here the Windrose Trajectory Model (WTM) (Trukenmüller and Friedrich, 1995) has been used to estimate the concentration and deposition of acid species. WTM is a user-configurable Lagrangian trajectory model, derived from the Harwell Trajectory model (Derwent and

Nodop, 1986); it differentiates between 24 sectors of the wind rose, such that from each sector a straight-line trajectory arrives at the receptor point. Concentrations at the receptor point are obtained by averaging over the results from these trajectories, suitably weighted by the winds in each 15° sector.

The creation of ozone has been modeled with the Source-Receptor Ozone Model (SROM) which is based on source-receptor (S-R) relationships from the EMEP MSC-W oxidant model for five years of meteorology (Simpson et al., 1992). Input to SROM are national annual NO_x and anthropogenic NMVOC emissions data from 37 European countries, while output is calculated for individual EMEP 150x150 km2 grid squares by employing country-to-grid square matrices. To account for the non-linear nature of ozone creation, SROM utilises an interpolation procedure allowing S-R relationships to vary depending upon the emission level of the country concerned (Simpson and Eliassen, 1997).

The ECOSENSE model (Krewitt et al., 1995), an integrated impact assessment model developed within ExternE, combines the results from the atmospheric dispersion models and the databases covering receptor data (population, land use, agricultural production, buildings and materials, etc.), meteorological data and emission data for the whole of Europe. Together with dose-response functions and monetary values stored in EcoSense, physical impacts and resulting (marginal) damage costs have been calculated within a consistent modeling framework, taking into account the information on receptor distribution. Impacts due to a point or line emission source are taken into account on a European scale, i.e. the dispersion of pollutants and related impacts are followed up throughout Europe.

Several validation tests have also been carried out to confirm the accuracy of the results. For example, the consistency between ISC and ROADPOL has been checked, and the concentrations predicted by WTM have been compared with measured data and with calculations of the EMEP program, the official program for the modeling of acid rain in Europe.

Health Impacts

The concentration-response (CR) functions for health used by ExternE are assumed to be linear (without threshold). Note that for the calculation of incremental damage costs there is no difference between the linear and the hockey stick function (with the same slope), if the background concentration is everywhere above this threshold; only the slope matters. For the classical air pollutants (particles, NO_x, SO_2, O_3, CO) there is some evidence of linearity down to levels as low as the background levels in most industrialized countries; the precise form of the CR function at extremely low doses is irrelevant for these pollutants.

In ExternE the working hypothesis has been to use the CR functions for particles and for O_3 as basis. Effects of NO_x and SO_2 are assumed to be subsidiary. The principal effects of NO_x and SO_2 arise indirectly from the particulate nature of nitrate and sulfate aerosols (NO_x is also a precursor for ozone) and they are calculated by applying the particle CR functions to these secondary aerosol concentrations. With this assumption the impacts of NO_x and SO_2 per kg of pollutant are roughly comparable to PM_{10}. But the uncertainties are large because there is insufficient evidence for the health impacts of the individual components or characteristics (acidity, solubility, …) of particulate air pollution. In particular there is a lack of epidemiological studies of nitrate aerosols because until recently this pollutant has not been monitored by air pollution monitoring stations. A summary of the most important CR functions for PM is shown in Table A1, together with the monetary values.

The exact functions being used are now old and in some respects out-of-date; we have plans to revise them in the next couple of years. However, the impact pathways being quantified remain generally valid – there are some new pathways, such as particles and infant mortality – and for existing pathways, newer evidence will improve but not markedly change the results. More evidence and improved understanding will, however, have reduced somewhat the uncertainties associated with the ExternE quantifications.

Table A1. C-R functions and costs for PM_{10}, as adapted and recommended by ExternE (1998). The exposure response slope, f_{CR}, has been expressed in units of cases/(person·yr·µg/m^3), relative to average population (thus it includes the fraction of the population that is affected).

End point for PM_{10} and reference	**f_{CR}** cases per (pers·yr·µg/m^3)	**Cost per case** €/case	**Cost per person** € per (pers·yr·µg/m^3)	**%** of PM_{10} cost
Chronic Mortality YOLL (Pope et al., 1995)	4.10E-04	84330	3.46E+01	85.0%
CB, Adults (Abbey et al., 1995)	3.92E-05	105000	4.12E+00	10.1%
RAD, Adults (Ostro, 1987)	2.00E-02	75	1.50E+00	3.7%
Bronchodilator usage, Asthmatic adults (Dusseldorp et al., 1995)	4.56E-03	37	1.69E-01	0.4%
Chronic cough, children (Dockery et al., 1989)	4.14E-04	225	9.32E-02	0.2%
CB, children (Dockery et al., 1989)	3.22E-04	225	7.25E-02	0.2%
HA, Cerebrovascular (Wordley et al., 1997)	5.04E-06	7870	3.97E-02	0.1%
Cough, Asthmatic adults (Dusseldorp et al., 1995)	4.69E-03	7	3.28E-02	0.1%
Congestive heart failure, Asthmatic 65+ (Schwartz and Morris, 1995)	2.59E-06	7870	2.04E-02	0.1%
Bronchodilator usage, Asthmatic children (Roemer et al. 1993)	5.43E-04	37	2.01E-02	0.0%
HA, Respiratory (Dab et al., 1996)	2.07E-06	7870	1.63E-02	0.0%
LRS, Asthmatic adults (Dusseldorp et al., 1995)	1.70E-03	7.5	1.28E-02	0.0%
Cough, Asthmatic children (Pope and Dockery, 1992)	9.34E-04	7	6.54E-03	0.0%
LRS, Asthmatic children (Roemer et al., 1993)	7.20E-04	7.5	5.40E-03	0.0%
Total PM_{10}			**4.07E+01**	100.0%

HA = hospital admission; CB = chronic bronchitis; LRS = lower respiratory symptoms;
RAD = restricted activity day; YOLL = years of life lost.
To derive f_{CR} from the data in the references (given e.g. as % increase per receptor), we have assumed:
3.5% of population is asthmatic, children are 20% of population, 14% of population is over 65.
For chronic mortality f_{CR} has been obtained by integration over life tables (ExternE 1998) or the Gompertz function for age-specific mortality (Leksell and Rabl, 2001), assuming that it applies only to the population over 30 years (= cohorts in Pope et al. (1995)).

Appendix B. Values of a Life Year Implicit in Public Decisions

Data on the costs of risk-reducing measures in the USA have been collected in an interesting study by the Harvard Center for Risk Analysis (Tengs et al., 1995). More than 500 life saving interventions were identified and the implied value of a YOLL was determined. The results show that there is an enormous range of cost/YOLL values, spanning over 11 orders of magnitude; such variations between the costs of different interventions were found in almost every category. The cost-effectiveness varies between different sectors of society, as can be seen from Table B1 which summarizes the median of the cost/YOLL estimates. The median cost is especially high in the environmental domain, $4,200,000, far higher than the median of $19,000 in the health care sector.

Table B1. Median of cost/YOLL estimates as a function of sector of society and type of intervention in the USA. Adapted from Tengs et al. (1995).

	Type of intervention			
Sector of society	**Medicine**	**Fatal injury reduction**	**Toxin control**	**All**
Health care	$19,000	na	na	$19,000
Residential	na	$36,000	na	$36,000
Transportation	na	$56,000	na	$56,000
Occupational	na	$68,000	$1,400,000	$350,000
Environmental	na	na	$4,200,000	$4,200,000
All	$19,000	$48,000	$2,800,000	$42,000

na = not applicable

An analogous study, following the same methodology and analyzing over 150 interventions, was carried out in Sweden by Ramsberg and Sjöberg (1997). Results are summarized in Table B2. Most of these interventions are implemented, and practically all have other objectives in addition to saving lives. The authors compare their results with those of Tengs et al., see Table B3. For medicine and fatal injury reduction the costs are comparable (approximately $20,000 in Sweden and $40,000 in the USA), but for toxin control the median cost is two orders of magnitude higher in the USA than in Sweden.

Table B2. Results for mean and median of cost/YOLL estimates in Sweden, in 1993$. Adapted from Ramsberg and Sjöberg (1997).

Category	**n**	**Mean**	**Median**
Medicine	101	$1,240,000	$14,000
Radiation	13	$30,000	$1,400
Road safety	32	$242,000	$66,500
Life style risks	3	$470	$340
Fire protection	7	$211,000	$15,000
Electrical safety	2	$1,245,000	$1,245,000
Accidents	1	$280,000	$280,000
Environmental pollutants	5	$235,000	$235,000
Crime	1	$15,000	$15,000

Table B3. Comparison of median cost/YOLL between Sweden and USA. Adapted from Ramsberg and Sjöberg (1997).

	Median cost/YOLS	
	Sweden	USA (Tengs et al. 1995)
Medicine	$13,800	$19,000
Toxin control	$19,600	$2,800,000
Fatal injury reduction	$69,000	$48,000
All	$19,500	$42,000

For the purpose of determining a value that could be recommended as guideline for environmental policy, it is not the median in these tables that is relevant but rather the upper range of values for interventions that are actually implemented. Data for the USA are difficult to interpret in this sense because the range is very large, covering many orders of magnitude. For Sweden Ramsberg and Sjöberg say that most of the interventions they consider are implemented, but even here the range is so wide that it is difficult to extract a recommendation. In any case it seems that the value of 84,000 €/YOLL chosen by ExternE (1998) is compatible with the numbers in Tables B1 to B3.

Appendix C. Derivation of Life Quality Index

The general idea is that a person's enjoyment of life, or utility in an economic sense arises from a continuous stream of resources available for consumption over the entire life. Therefore income required to support consumption and the time to enjoy are two determinants of the life quality. For a person at age a, the lifetime utility can therefore be interpreted as total consumption incurred over the remaining lifetime, which is a random variable.

The mathematical derivation is briefly described here and details can be found elsewhere (Pandey and Nathwani, 2003). Denote the consumption rate at some age τ as $c(\tau)$ (\$/year), and assume that a valid function, $u[c(\tau)]$, exists that can quantify the utility derived from consumption. The probability of survival in the period a to t is denoted by $S(a, t)$. The present value of life-time utility for a person is equivalent to integration of $u[c(\tau)]$ from the present age a till a terminal age T with a suitable discount rate to reflect the fact that individuals tend to undervalue a prospect of future consumption in comparison to that of present. Thus,

$$L(a) = \frac{1}{S(0,a)} \int_a^T S(a,t) u[c(t)] e^{-r(t-a)} dt \qquad \text{(A1)}$$

where r denotes the rate of time preference for consumption. Assuming a power utility function and constant consumption rate, i.e., $c(t) = c$, and $u(c) = c^q$, eqn.A1 can be written in a compact form as

$$L(a) = u(c)e(a) = c^q\, e(a) \qquad \text{(A2)}$$

The life-time utility, $L(a)$, is a surrogate measure of quality-of-life of a person of age a. This type of reasoning primarily originates from the fundamental work of Usher (1973) on the impact of historical improvement of LE on economic growth.

The life-quality at the societal level is an aggregate of the values for all individuals in the society. To achieve this, $L(a)$, should now be integrated over the distributions of population age and consumption rate. As a matter of simplification, we assume that the consumption rate is equivalent to the real gross domestic product per person per year (G), a valid measure of average consumption in society. Integrating $L(a)$ over the population age-distribution, $f(a)$, leads to

$$LQI = \int_0^T L(a) f(a) da = c^q \int_0^T e(a) f(a) da = G^q E \qquad \text{(A3)}$$

where E denotes the discounted life expectancy averaged over the age-distribution of the population. The exponent q can be shown to equal the ratio $q = w/(1-w)$ where w is the average fraction of time spent on work in a country for producing G (11). For industrialized nations a typical value of q is 0.15.

The societal life-quality function, LQI, is a utility function as well as a composite social indicator, since it consists of two important indicators of development, namely GDP per capita and life expectancy. By setting E equal to LE at birth and ignoring the discounting, LQI was used to rank the level of national development (Nathwani et al., 1997) similar to Human Development Index proposed by the United Nations Development Program (UN, 1990).