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Working Paper 1998:13

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A methodology for integrated assessment applied to a case study in Hungary

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ISSN: 0804-452X

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Dr. scient. Thesis to the Department of Chemistry, Faculty of Mathematics and Natural Sciences, University of Oslo, 1998

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List of papers

Paper 1
Aunan, K., Seip, H.M., and Aaheim, H.A., 1995. A model framework for ranking of measures to reduce air pollution with a focus on damage assessment. In: Hargittai, I. and Vidóczy, T. (eds.), *Combustion Efficiency and Air Quality*. Plenum Press, New York, pp. 269-283.

Paper 2
Aunan K., 1996. Exposure-response functions for health effects of air pollutants based on epidemiological findings. *Risk Analysis* **16**(5), 693-709.

Paper 3
Aunan, K., Pátzay, G., Aaheim, H.A., and Seip, H.M., 1998. Health and environmental benefits from the implementation of an energy saving program in Hungary. *Science of the Total Environment* **212,** 245-268.

Contents

1. Introduction

Environmental problems have different characteristics in terms of space and time. Noise, for instance, affects only those being close to the source, and the problem is instantly solved for these if the source is removed. Health effects resulting from local sources of air pollutants may disappear very soon after the sources are controlled, but for some diseases an enhanced frequency may be seen in the affected population for years and even decades. Acidification of soil and water resources and increased level of tropospheric ozone are the accumulated results of long-range transboundary pollution taking place over several years, and if abatement measures are implemented today the systems affected may not restore their original state in a long time. At the end of the temporal and spatial scales are the issues of stratospheric ozone layer depletion and climate change, problems that have not been realised until rather recently, but which are results of human activities taking place over a long period of time. Typically, the complexity and unpredictability increase along the axes, making both the science behind the problem identification and the abatement planning heavily burdened with uncertainty. The spatial and temporal characteristics of the problem in question imply that widely different policy response strategies may seem required.

To a large extent, the main environmental problems that cause concern today are due to the same emission sources. This fact, together with an increasing focus on cost-effectiveness in abatement planning, has created a rapidly increasing interest in methodologies for undertaking "integrated assessments" (IA) over the past decade. The term, however, is used for very different analyses, depending on what are being integrated. As pollution of the environment may be caused by economic activities, as well as the pollution causes some socio-economic damage, the term IA may be used for studies of these vertical inter-linkages for one specific environmental problem. For instance, several attempts have been made to model the mutual impacts of economic activities and various climate change effects (as for instance sea level rise) on a global level or for large regions of the world (see e.g. Dowlatabadi (1995) for an overview of some major policy motivated models and Shlyakhter et al. (1995) for a discussion of some main factors to be considered in integrated analyses of global climate change). The RAINS model for Europe (Alcamo et al., 1990) and the RAINS-Asia model (Downing et al., 1997) have been developed to analyse trends in emissions, estimate regional impacts of resulting acid deposition levels, and to evaluate costs and effectiveness of alternative mitigation options. These models thus integrate the cause-effect chain and the policy response issues related to acidification for two large regions of the world. Recently, work has been initiated to design a methodology for integrated assessment of mitigation policies concerning NO_x and VOC and their effect on acidification and ozone levels in Europe (see Simpson and Eliassen, 1997).

The climate change issue is one of the most important environmental problems today. According to the International Panel on Climate Change (IPCC) the "balance of evidence suggests that there is already a discernible human influence on global climate" (Houghton et al., 1996). A further increase in the atmospheric CO_2 concentration seems inevitable in view of the present dependency on fossil fuels in the western countries and the rapid economic growth in China and some other developing countries. It is therefore of great importance to find strategies in dealing with the climate change problem. However, the advice given in a large part of the economic literature, based on integrated assessment models as mentioned above, is that only moderate actions are recommended at present (see e.g. Nordhaus, 1993). There are, at least, two major reasons why this may be a poor advice.

Firstly, until now most assessments of the costs and benefits of curbing greenhouse gas (GHG) emissions have not taken into account the *secondary benefits* of mitigation measures (also called ancillary benefits in the literature). These includes *i.a.* the benefits that are gained due to the fact that many of the relevant measures also curb the emissions of air pollutants[1]. This issue has got surprisingly little attention in the literature and in the public debate. However, there is an increasing interest in the area, as shown by works by for instance Ayres and Walter (1991), Barker (1993), Alfsen and coworkers (1992; 1995; 1997), Munn (1995), and Burtraw and Toman (1997). It is also pointed at as an important element by the Intergovernmental Panel on Climate Change (Bruce et al., 1996).

Secondly, the choice of discount rate in the analyses has a profound effect on the present value of environmental effects and mitigation costs. The traditional economic tools and approaches for cost benefit analyses may simply be inadequate for analysing effects that appear decades after the investments in abatement measures were made. The questions of discounting far-future environmental benefits and dealing with values that might change over time have become subject to extensive discussions within the field of environmental economics the last years (see e.g. Nordhaus, 1997; EC, 1995a).

The aim of the work described in the following is to contribute to methodologies that may be useful in promoting a comprehensive environmental decision making, by within a limited geographic area *integrating the damage assessment across some major environmental problems*. The methodology is applied to a case study in Hungary (the CAPE project – 'Climate, Air Pollution and Energy: Cost-Effective Strategies for Reduction of Emissions'), where the benefits of implementing an energy saving program are assessed in terms of reduced physical damage and the entailed economic benefits for the society. The work is closely linked to other work at CICERO concerning the issues of discount rate and macroeconomic modelling of secondary benefits of GHG reductions (see e.g. Aaheim et al., 1997).

The study is basically a bottom-up cost benefit analysis, as described in paper 1 and 3. Figure 1 in Paper 1 outlines the steps in the analysis. The approach resembles what has been called the *impact pathway* or the *damage function approach* for instance in the joint CEC/US fuel cycle cost study, ExternE (EC, 1995a) and in the Norwegian project LEVE (see e.g. Haagenrud and Henriksen, 1995). At the outset the project intended also to integrate those political science issues regarded as important for what kind of abatement policy that was worthwhile looking at in an Hungarian case study. In the first report from the project policy implications of the Hungarian situation being a country in economic and political transition are discussed (Seip et al., 1995). However, this part has not been followed up, and the fact that we analysed an energy saving program that was already elaborated and proposed by the Hungarian authorities, to some extent reduced the relevance of an explicit political analysis.

[1] For convenience the term 'air pollutant' is in the following restricted to those components having an adverse effect on health and environment at concentration levels found in ambient air, and GHGs are not included in the term. In many cases it may be more natural to treat reductions in air pollutants as primary, and to view the entailed reduction in GHG emissions as a secondary, 'spill-over' effect.

2. Decision frameworks

Any rational decision process implies weighing of pros and cons of alternative choices leading to a recommendation of the alternative that altogether seems to meet the main objectives in the optimal way. To be able to assess the advantages, or benefits, of implementing pollution abatement measures, knowledge of the different steps in the impact pathway is required. These are *i.a.* the composition of emissions, how these are spread, transformed and deposited, and finally the severity, magnitude and distribution of the damage incurred due to the emissions.

It may be useful to take a brief look at different frameworks for setting goals and making decisions within the field of environmental protection, in order to view the cost- benefit approach in a broader context. To what extent quantitative estimations of exposure-response relationships and economic valuation of the damage, the core elements in the CAPE study, actually are needed may vary among the different approaches. The various decision frameworks imply diverse guidelines for decision-makers, hence using different rules in different situations may lead to inconsistency in the environmental legislation. In practice one usually finds a combination of several principles that can be formulated in a number of ways. Morgan (1993) uses the following categorisation:

- technology based
- rights based
- utility based

The first principle prescribes obligatory use of the best available technology (BAT) at the given point of time. This implies that the target changes with the technological development. In some cases this principle appears to be rather defensive, in other cases the contrary, e.g. when the emission source in question obviously does not contribute significantly to a given pollution problem. Basically, the BAT-principle does not address questions of cost efficiency and equity, but solely focus on the status of available technology. However, the implementation costs are important when it comes to the concrete definition of what is ‘best available’, a concern giving rise to the more recent notion BATNEEC, Best Available Techniques (both technology and operating practices) Not Entailing Excessive Costs (see e.g. Carlyle, 1996). The definition of ‘excessive’ is, however, still left to the faculty of judgement.

The BAT-principle, combined with a fixed percentage goal for reductions, was an important element in the first generations of international agreements on reduction of *i.a.* SO_2 and NOx. This type of agreement has no strong theoretical platform, strictly speaking, but has been employed due to its virtue of being a simple and pragmatic approach. In the second generation of the SO_2- and NO_x-agreements, however, the principles of critical levels/loads and target levels/loads have become central elements.

Thus, the BAT approach is largely independent of explicit calculations of risks connected to the different hazards and masks the inevitable value judgements and social trade-offs between risks and costs.

In Morgans terminology, the rights-based principle places justice in front of utility. This implies that decision-makers should consider the situation for vulnerable individuals and recipients simply because they have a right to protection from harm. With this approach, the most important basis for decisions is knowledge about critical levels or critical loads, guidelines based on LOAEL (lowest observable adverse effect level) and estimates of acceptable doses. For risk of fatality a *de minimis* level is often set at what would cause less than a 10^{-6} increase in a persons average annual probability. The inevitable dilemma arises, however, because no such standard based on some average annual probability, is able to provide equal protection to all citizens (Shrader-Frechette, 1991).

The intention of protecting sensitive groups in the population, vulnerable vegetation, landscapes etc. is usually articulated in environmental legislation. However, one hardly finds the rights-based approach in a pure form. In the real world there are budget restrictions, hence some kind of utilitarian approach is employed. The utility-based approach has its origin in economic theory, where the principle of optimal allocation of resources and maximising the total welfare in the society is central. Two main types of amenities are vital to the social welfare, private goods and public goods. The environment in which we live is a public good and pollution may deteriorate it. Additionally, pollution may also reduce private goods, for example if high ozone levels damage agricultural crops and thereby reduce the income of the farmer. In traditional cost/benefit analysis one attempts to evaluate all the costs and benefits of specific projects on behalf of the society, and to quantify them in one unit (usually money). The economic paradigm thus implies that the focus is taken away from the individual risk of damage or suffering, onto the aggregate damage to the affected population. The interest for employing economic damage estimates in air pollution regulation has increased, and in the USA the claim for economic estimates of the environmental consequences of larger projects is obligatory. In other countries, including Norway, this has until recently been more ad hoc and common guidelines and incentives for evaluation of environmental impacts do not exist. The concept is, however, gradually gaining more attention (Markandya, 1993; Kopp et al., 1997; Navrud and Pruckner, 1997).

At least in its original form, cost-benefit analysis says that one should choose the action with the best expected value or utility (the *Bayesian decision criterion*). The expected value or utility is defined as the weighted sum of all possible consequences, and where the weights are given by the probabilities associated with each consequence. This clearly contrasts the *maximin decision rule* advocating the rationality in choosing the action that avoids the worst possible consequences, and also decision rules emphasising equity and distribution consideration one way or the other, e.g the Paretian criterion (see Shrader-Frechette, 1991). Nevertheless, as we shall see later on, the outcome of cost-benefit analyses may be useful also in the context of following the maximin criterion, provided that the full uncertainty distribution for the outcome is given. A prerequisite is, of course, that the probability distribution can be estimated at any reasonable level of certainty for all important parameters and variables.

Different decision frameworks have different qualities in terms of economic efficiency, distributional equity and administrative simplicity, which are important criteria for evaluating public policy. According to Pearce et al. (1992) there is an antagonism between these qualities. Whereas simple technology-based approaches (e.g. emission standards) have a very high administrative simplicity, they have a low score on economic efficiency and equity. Regulations and bans aiming at ‘no risk’ or a lowest possible risk for vulnerable groups, also have a low economic efficiency, but have a higher score on equity. The cost-benefit (or risk-benefit) approach has a high degree of economic efficiency, but it may suffer from a low administrative simplicity and equity. However, whether this is the case depends on how the authorities perform such analyses.

3. Quantitative risk assessment

As environmental policy has got increasingly more dependent on making explicit scenarios for changes in environmental quality and damage, so has the need for quantitative calculation of risk[2] of adverse effects increased. Let us very briefly recapitulate the history of risk assessment, focusing health risks.

As early as the 1st century B.C. the Greeks and Romans had become aware of the adverse effects of exposure to lead through various mediums. Modern risk analysis, however, has its twin roots in mathematical theories of probability and in scientific methods for identifying causal links between adverse effects and different types of hazardous activities. An early prototype of modern risk analysis was La Place's estimation, in the 17th century, of how vaccination could reduce the probability of dying of smallpox (see Covello and Mumpower (1985) and references therein).

Calculation of different kinds of environmental risks is an area where the research accelerated in the 1970s. Quantitative risk assessment gradually became an independent scientific discipline, with USA as a pioneer country. By means of analytical methods like probabilistic assessments one set out to find the 'real risk' connected to different hazards/technologies. The work partly had its motivation in the need for protecting the population and environment and partly in the needs of industrial and economical interests claiming that the constantly more stringent environmental legislation was inconsistent, contradictory and counterproductive (Misa, 1990).

However, a certain scepticism evolved in the public opinion towards the risk analyses of experts and authorities, as expressions like 'risk technocrats' indicate. There were many reasons for this. For instance, the early generations of risk analyses often focused solely on mortality, or lost years of life, while people in general also are concerned about morbidity. From a socio-economical point of view morbidity is important as well and should be incorporated in the analyses. Moreover, in connection to the operation of nuclear power plants, it is a fact that even though the risk of technical failure may be estimated fairly good, the behaviour of the operating personnel can not.

Traditional quantitative risk assessment regarding health effects of chemicals mainly build on toxicological studies. It is very difficult to use epidemiological data for verification since toxicological studies generally cannot cope with complex exposure situations and synergism between components (Silbergeld, 1993). Increasingly, therefore, one started to use results from epidemiological studies directly in the assessments, because epidemiology provides the only direct source of information. Particularly, this is the case when it comes to long-term cumulative effects.

A fundamental problem in basing decisions on quantitative risk assessments is that it is not exclusively the 'real risk' that matters to people. Attitudes towards different risks are not solely determined by expected numbers of deaths and people becoming ill per unit of time. Research indicates that attitudes towards risks are influenced for instance by how well the

[2] Risk refers to the individual probability of being affected, or the collective risk in a population (or a sub-population) which is the average frequency of affected people.

causal connection is understood, in what way mortality and morbidity manifest (the ‘degree of dreadfulness’), the possibilities one has to control the exposure to the hazard and whether exposure is voluntary or not (i.e smoking versus air pollution) (see *i.a.* Morgan, 1993 and Slovic, 1987)[3]. For instance, there are indications that people have considerably higher willingness to pay for reducing the risk of dying of cancer than dying in a traffic accident (Åkerman, 1987). If the effort that society should devote to reducing health risks was governed solely by the size of the risks, preventing smoking should certainly be given a much higher priority than ambient air pollution prevention.

Whereas some experts may complain of laymen's lack of knowledge and irrationality concerning perception of risk, others argue that the presumably objective risk analysis is non-operational when it comes to analysis of action strategies and policy analysis (Silbergeld, 1993). If the perception of risk in some cases is ‘wrong’, according to the experts' calculations of probabilities and extent of damage, this may reflect that risk connected to this particular type of environmental hazard is especially unwanted. That is, minimising this risk is highly evaluated. In a democratic society it would be unsound to brush aside attitudes in the opinion as irrational, because they may represent an ethical dimension that the scientific risk analyses have a tendency to disregard. This may consist of several elements, for instance attitudes towards uncertainty, and considerations of equity and distribution. As we shall see in the following, the various methods of adhering economic values to environmental amenities actually are attempts to reveal people’s attitudes, or preferences, towards these issues. As an example, contingent valuation studies have showed that respondents may state a higher willingness to pay for avoiding health effects in children than in themselves (Navrud, 1997).

[3] There is a huge literature about paradoxes showing that people make choices conflicting with the assumptions on which the theory of expected utility is built, read e.g. about the Allais paradox in Aaheim (1992).

4. The physical impact assessment

4.1. Population exposure assessment

Exposure to a pollutant is defined as the event when a person comes into contact with the pollutant of a certain concentration during a certain period of time (WHO, 1987). Air pollution levels show substantial temporal and spatial variations. As noted in Paper 2, there is generally no consensus regarding what averaging times are appropriate for studying and predicting various health effects of air pollution. Concerning acute effects associated with ozone exposure, peak values, indicated e.g. by the 1-hr daily maximum values, are used in several epidemiological studies reporting an association (see for instance Burnett et al., 1994). However, peak values are often highly correlated both with 8-hr mean and daily mean values, and associations are found also in studies using these averaging times (see EC, 1995b for a review of ozone epidemiology). A corresponding situation also applies to chronic diseases associated with air pollution. As discussed in Paper 2, it is in itself difficult to demonstrate the risk of cumulative effects due to long-term exposure, and additionally difficult to find an 'optimal' averaging time and period (e.g. childhood, recent years or cumulative lifetime) to use in the studies of the various effects. In the studies reviewed in Paper 2 both annual mean and 5-y mean were used as independent variables.

Ideally, exposure should be assessed by considering the full time series of concentrations, C, measured by a device attached to the person being studied. The concept of integrated personal exposure *(E)* takes into account the temporal and spatial variation in air pollution level in the following way (CEC, 1992):

$$E = \int_{t1}^{t2} C(t).\,dt$$

which can be approximated and rewritten for J different micro-environments that are visited over a certain period of time t_{ij} by individual i as:

$$E_i = \sum_{j}^{J} C_{ij} \cdot t_{ij}$$

There can be significant differences in the time-activity pattern between individuals and between subgroups in a population. The time spent outdoors is, generally, considerably less than the time spent indoors, even for groups having a higher outdoor/indoor time-activity ratio than average. On the average for populations as a whole, there are, however, great similarities in time-activity patterns across western industrial countries (CEC, 1992; Clayton et al., 1993). As discussed in Paper 2, the epidemiological studies being the basis for our estimates of health effects, represent random cross-sections of populations regarding the personal exposure to outdoor concentration ratio. This makes the application of the exposure-response functions (or rather concentration-response functions) in combination with data or estimates of ambient air concentration levels in a case study, a feasible method for quantitative assessment given limited research resources. The price to pay is that one does not get hold of the possible disparity in exposure and response among subgroups in the population. The relatively greater vulnerability of some population groups, however, is relevant to any discussion of the equity case for reducing air pollution.

Relying on outdoor concentration data may in some cases be a bewildering and not particularly reassuring approach, in epidemiological research as well as in assessments of the possible benefits of reducing emissions. Concerning the Hungarian case study of benefits of an energy saving program the quality of the monitoring data that were available is discussed by Kruse (1995) and Seip et al. (1995). The situation in Hungary adds to the general recognition that there is a great need for standardising the methods for monitoring ambient pollution levels. This would be beneficial not at least in the context of assessing population exposure, e.g. in order to facilitate comparisons between populations (see Kuchuk et al., 1995).

We decided to use the data from the continuous network in Budapest to establish a function that could be used to estimate the PM_{10} level in the 80 Hungarian cities that were included in the analysis (PM_{10} is the pollution component most important for the health effect assessment made here, See Paper 2). As described in Aunan et al. (1997), the tentative test of this procedure indicated that for our purpose, which was to study the effect of an energy saving program, a reasonable approximation could be made.

Figure 1 shows the population exposure distribution resulting from the procedure. For Budapest, by far the biggest city, having more than 2 million inhabitants, we were able to estimate exposure on a district level (34 districts/sub-districts). For the other cities, however, the exposure assessment is an average for the city or town. The distribution resembles, as expected, a log-normal distribution[4].

As a comparison, recent data from US cities rarely show annual average PM_{10} concentrations over 50 $\mu g/m^3$, and in the Dutch National Monitoring Network the annual average levels were in the order of 40 $\mu g/m^3$. In other areas in Central and Eastern Europe, as well as in urban areas in China and India, annual averages may significantly exceed 100 $\mu g/m^3$. In central areas of Katowice in Poland, for instance, the annual average PM_{10} level is approximately 150 $\mu g/m^3$ (WHO, 1995; Saksena et al., 1992; Cropper et al., 1997b; RIVM, 1995).

Concerning the exposure assessment for subgroups, as for instance asthmatics, children and elderly, we assumed that they were subject to the average exposure level. As discussed above, this may to some extent have led to a misclassification of exposure for some groups. Generally, however, the possible errors in exposure assessments resulting from using average ambient air concentrations are probable much larger for populations being exposed to very high levels of indoor pollution, which typically is the case in areas in China and India and other developing countries. For instance, the use of bio-fuels or coke in un-vented cook-stoves in these countries, implies that especially women, but also children, incur an exposure to particulate pollution that is considerably higher than average for the population (Tata Energy Research Institute, 1995; Saksena et al., 1992; Smith et al. 1994).

[4] More details on population exposure in Budapest are given in Aunan and Seip (1997).

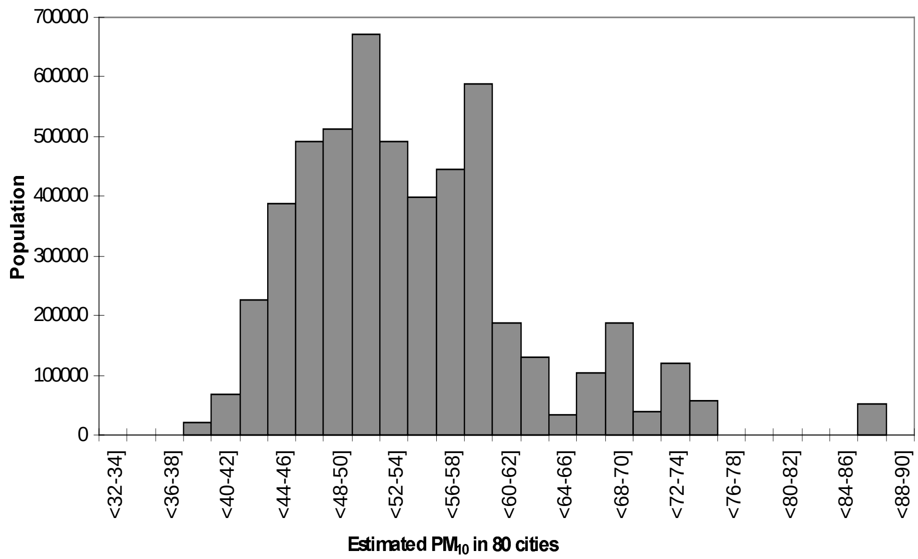

Figure 1: Population-exposure distribution. Annual average PM_{10} -levels estimated from monitored NO_2 –levels in Hungarian cities.

4.2. Exposure assessment for material damage and cereal crop loss

Kruse (1995) estimated the annual saving in total corrosion costs in Budapest if the average SO_2-levels were reduced to less than 20 $\mu g/m^3$, and found it to be about 100 mill. US$. In the case study described in this thesis the annual saving resulting from implementing an energy saving program was assessed. As seen from Table 1 in Paper 3 an SO_2-reduction of about 6% is estimated for the program. According to our calculations a flat reduction must be in the range 25-35% in order to secure that the annual average SO_2 level is below 20 $\mu g/m^3$ in all districts.

Figure 2 shows the SO_2 level in some districts as a function of the overall reduction in SO_2 emissions. In the calculation of the benefit of the energy saving program the building density in the districts was used. (As discussed in Kruse (1995) there are indications that the monitoring data that were available for the whole city to some extent were too low).

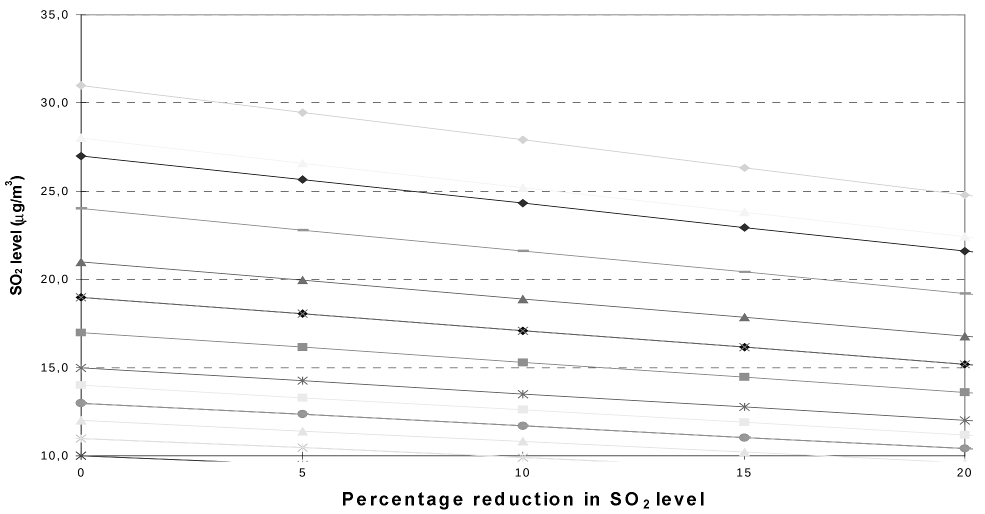

Figure 2: SO_2-level in the districts (or sub-districts) in Budapest that have the highest level (figures from 1992), estimated as a function of the overall percentage reduction in the SO_2 level.

The exposure assessment for cereal crops in Hungary was based on the calculated AOT40 values (accumulated exposure over the threshold of 40 ppb – unit: ppb-h) given by EMEP (EMEP/MSC-W, 1996). The EMEP grid cells are 150 km x 150 km, and the distribution of the areas for cereal crop production was not investigated, hence the exposure assessment in our study is rather rough. Factors that may lead to deviations from the average AOT40 values are mountainous topography (AOT40 generally increases with height) and inhomogeneous distribution of emission sources of ozone precursors (Spangl, 1996). For instance in the county Borsod-Abaúj-Semplén in Hungary much industry is situated near the Carpathians. These areas are, however, not important to agricultural production.

In general the forest condition in Hungary is fairly good and has been stable for many years (UN-ECE/CEC, 1996). We have therefore not estimated possible improvements that might follow implementation of the energy saving program. However, Hungarian scientists have observed serious problems in the forests in limited areas (the north-eastern mountainous area), as described in Seip et al. (1995).

4.3. Exposure-response functions

Exposure-response functions give the continuous relationship between the exposure level and the frequency of a specific effect in a receptor population or the recipient. However, only for some pollution problems knowledge about exposure-response relationships is at a level justifying use in quantitative estimations. For others more semi-quantitative methods must be used, for instance estimations of how much guidelines for exposure levels are exceeded. Briefly, the following methods are available in the assessment of effects of some major types of environmental pollution:

Human health:
- Air quality guidelines and exposure-response functions for air pollutants

- Drinking water quality guidelines
- Guidelines for tolerable daily/weekly intake of contaminated food
- Noise level recommendations/guidelines

Vegetation:
- Critical levels and critical loads
- Exposure-response functions (for ozone)

Materials:
- Exposure-response functions

In the Hungarian case study the focus was on the reduced air pollution that could be obtained by reducing the energy use. Reduced damage to health, materials and crops were considered the most important effects. Exposure-response functions were used to assess the benefits. Based on a review of epidemiological literature functions for health effects were proposed within the study (see Paper 2). For crops and materials functions from other literature were employed.

There is today a large international literature on the effects of air pollution on human health. Whereas there is no longer doubt that effects occur, questions of the quantitative extent of the effects remain. However, an increasing number of clinical and epidemiological studies have found significant exposure-response relations for different types of health effects. In the context of environmental impact assessment and policy making, results from epidemiological studies are advantageous in many respects, as argued in Paper 2. The methodological development the last 10-15 years has also increased the reliability of these studies.

The exposure-response functions proposed in Paper 2, and employed in the case study in Hungary, apply to biological end-points solely (as opposed to consequential (or social) end-points as occupational absenteeism and hospital admissions which are end-points that are consequentially related to the biological end-points). Although probably better than for consequential end-points, the transferability of functions for biological end-point does, however, vary. Generally, functions from ecological studies (using statistics for whole populations) are more easily transferred as compared to e.g. functions from cohort studies or other types of panel studies, because the latter may, for one reason or the other, represent a biased sample. In any case, a reasonable similarity between the population in the epidemiological study and the population in the applied study, as concerns e.g. demography and background morbidity, is required. Problems related to transferability of the functions used in our study, which are based mainly on Western epidemiological studies, are briefly discussed in Paper 2 and Paper 3 (see also EC, 1995a, for a discussion on transferability of functions).

Probably, the most important health end-points are included in the health effect analysis made here. However, all effects are not described and some of the omitted end-points may be more important than the end-point pseudo croup which was included. For instance, functions for prevalence of headache and eye irritations are provided in the literature (see Aunan, 1995). Neither did we estimate asthma symptom days for children, although there is abundant evidence in the literature that air pollution contributes to an enhanced frequency of asthma episodes in children (see e.g. Berciano et al., 1989; Pönkä, 1991; Roemer et al., 1993; and

Walters et al., 1994). The reasons for these omissions were partly problems with deriving exposure-response functions from the studies, and partly the danger of double counting due to a possible overlap with the end-point chronic respiratory symptoms in children (CRS-Ch). In the study giving rise to the function for CRS-Ch, the smaller group of children that reported asthma or wheeze contributed to most of the cases of bronchitis that could be attributed to air pollution (see Paper 2). If a particular function for asthma days in children had been included, this would most likely have resulted in a partial double counting of effects. This also highlights the inherent problems entailed in the economic valuation of the effects: There is a willingness to pay for avoiding a disease *per se*, and in addition, there is a cost related to the days when the symptoms are worse and restricts activity more than usual (see also Section 5.1 and 5.2 below).

Implementation of the energy saving program will probably also lead to a modest reduction in the ozone level. There is a large literature indicating an association between various health effects and ozone exposure (EC, 1995b). Many of the end-points are the same as those found to be associated with particulate pollution and other air pollution components. As discussed in Paper 2 particles may be regarded as an indicator component in the exposure-response functions derived from the selected epidemiological studies (and applied in the case study). To some extent there would, again, be a danger of double counting if own functions for ozone had been applied. The synergism between air pollutants is, however, an essential feature that should be kept in mind, and remains an important challenge to further research.

To some extent energy saving in Hungary would also reduce the emission of heavy metals. In a study by Krupnick et al. (1996) in Central and Eastern Europe the health benefit related to reduced emissions of lead was estimated to be considerable in Hungary. However, large reductions have taken place since the data used in their study were collected, due to introduction of unleaded gasoline and a general lower content of lead since 1992 (Seip et al., 1995). Hence, much of the benefit estimated by Krupnick and co-workers has already being reaped by the Hungarians.

5. Assessment of the economic benefit from curtailing pollution

5.1. Methods of economic valuation of environmental goods

Valuation of environmental goods has become a large research field, and it is beyond the scope of this thesis to go detailed into it. Rather, the purpose of the parts of the study that concern economic valuation is to take some results provided by the economic literature and apply them to the output of our physical impact study. This gives an opportunity to compare the different types of benefits obtained from emission reductions. A sensitivity analysis is given in Section 6 and shows with all clearness that various approaches to put unit values to health end-points may lead to highly different outputs of our study. Also, taking a top-down approach will give a different output, as we shall see in the following (see also Aaheim et al., 1997). A judgement of which conceptual and methodological frameworks for evaluation are the 'better', is not made here. However, in order to put the resulting estimates into perspective let us briefly recapitulate some basic notions and have a brief look at the various approaches.

A basic idea of environmental economics is that to some extent society should accept a certain level of damage to human health and the environment, simply because of the usefulness of the activities that cause these damages. Thus, an important issue is to find what is the optimal level of abatement, beyond which further abatement efforts would 'cost more than it tastes'. The notion of *external costs, or negative externalities* is related to the damage that exceeds this optimal damage level. A negative externality may be defined to be the costs that fall on one group of people due to the activities of another group, and where the latter group does not fully compensate these costs.[5] Concerning electricity production, for instance, the negative externalities are the costs of the impacts of pollution not reflected in the market price. Imposing an environmental tax on the energy would then be an attempt to *internalise* these costs. Alternatively, one could establish regulatory standards in order to achieve the optimal emission level in a socio-economic sense. In both of these cases an internalised shadow price is imposed on the externality, and although there may still be some adverse environmental effects there is no external cost. This has the implication that measuring the environmental damage cost is not equivalent to identifying the external cost (see EC, 1995a).

There are principally two approaches to economic evaluation of environmental policy measures, described in Paper 1 and Paper 3 as the bottom-up approach (B-U) and the top-down approach (T-D). Concerning B-U analyses there are, again, principally two approaches to the estimation of the value of emission reductions. Taking health effects as an example, one approach is to estimate the costs that accrue due to lost productivity of the person that gets ill, medical cost, hospitalisation costs etc. These are often referred to as the damage cost, or cost of illness (COI) and can be estimated in various ways, see e.g. UNSO (1993). They rely on prices that can be observed in the market and are related to points at the *supply* function in a supply-demand diagram (see Figure 3, which we will come back to later).

[5] If the 'externality sufferers' *are* fully compensated, one may say that there is no 'relevant externality', or the externality is zero; i.e. the level of welfare is not changed. This is the *optimal* level of externality. A *positive externality* (an *external benefit*), on the other hand, is found in a corresponding situation where one agent generates a *positive change* in welfare for a third party (see e.g. Pearce and Turner, 1990, and EC, 1995a).

The other way is to estimate the willingness to pay (WTP) for a reduction in emission, or the compensation needed to accept the current level of emission, an approach that relates to the *demand* function (see Figure 3). Here, one can not easily rely on prices observed in the market. There are basically two approaches to estimate the WTP. The first is to get preferences *stated directly* by means of interview techniques. Such techniques are studies of lay peoples WTP in contingent valuation (CV) studies, and panel studies, which by use of interactive data programs assess the preferences in a small group of well informed persons. The other principal method is to *reveal* preferences *indirectly* by observing peoples behaviour for instance on the housing market (hedonic pricing techniques) (see *i.a.* Navrud, 1992; and Wenstøp et al., 1994).

The appropriateness of the willingness-to-pay approach has been intensively debated (see e.g. Arrow et al., 1993). A fundamental question is whether one gets hold of the *social values* by the methods mentioned above. For instance, is it possible by means of direct methods, to deduce how people will actually act, or is it rather their attitudes (to how one ideally should act) that is measured? Moreover, it may in some studies be unclear which considerations the interviewees included in the answers. For instance, for many health effects much of the costs (e.g. connected to medical treatment and hospitalisation) are carried by the society and not by the individual. It is then a question whether the costs not carried by the individual, and which magnitude may not be quite clear to the interviewees, are considered in the answer. The information given to the interviewees is thus of vital importance to the outcome of the study (see e.g. Navrud, 1992). The extent to which WTP estimates may overlap with damage cost estimates remains one of the unresolved methodological issues in the literature, and there are several obstacles to making general statements about a ratio between WTP and COI (the legitimacy of stating a general ratio is briefly discussed in Section 5.4). For health end-points WTP estimates are likely to overlap more with damage cost estimates in countries where the individuals pay more directly for health services. Besides, the overlap is likely to vary substantially between various end-points. For instance, the WTP/COI-ratio is likely to be much higher for mortality and chronic diseases, as compared to less severe end-points (US-EPA, 1995).

It is a question whether it poses an unrealistic cognitive demand upon people to ask them to state their preferences for environmental goods in monetary units (see *i.a.* Gregory et al. 1993, Kahneman and Knetsch, 1992). Fischhoff et al. (1988) argue that it may be wrong to assume as a premise that people do have a consistent set of values that ‘await quantification’. Especially when it comes to unfamiliar issues for which they have never thought through the implications of different values, elicitation procedures (e.g. by interactive techniques as mentioned above) may contribute to the ‘shaping’ of the values expressed. One pitfall is that a systematic error is induced due to a possible hinting of the ‘correct’ answers.

Another serious problem with the concept of estimating preferences in interview situations is the evidence for what Tversky and Kahneman (1981) has called ‘framing’. This relates to the presumption that if a choice is to be made from a set of alternatives, a rational choice should depend only on the membership in the set and not on how the alternatives are described. Unfortunately, several experiments have indicated that such invariances may not hold; the way the alternatives are described may have an impact on the choice.

In the work made in this thesis, we used the results from various WTP studies in the economic valuation of the health benefit, because it, after all, was regarded as a reasonably well-founded concept able to incorporate considerations of multiple dimensions. Also, empirical estimates were available in the literature. In principle there may also be a WTP for avoiding damage to agricultural production, which might not be the same as the economic costs of crop loss. And there is in Hungary for sure a WTP for avoiding damage to cultural/historical/architectonic interesting buildings etc., that adds to the replacement and maintenance costs. However, this approach has not been employed in the assessment of the economic value of these effects, where we relied on the damage costs that could be estimated from market prices.

5.2. Application of unit values for health effects in the case study

Due to the lack of valuation studies in Hungary, unit values derived from Western studies were adopted in Paper 3 to estimate the health benefit from implementing the energy saving program. The assumptions that were made and the problems that are related to transferring unit values are briefly stated. As little empirical evidence exists on the actual relationships between income level and WTP for environmental improvements in Hungary today, it is difficult to assess what is the better way to manage the issue of benefit transfer. As noted by Navrud and Pruckner (1997) uncritical transfers of CV results can provide invalid estimates and undermine the trust in non-market valuation work. They also suggest that benefit transfer is best suited for tasks where the need for accuracy is low, basing their view *i.a.* on the results of a test study concerning water quality improvements (Bergland et al., 1995). In the literature one finds various approaches to benefit transfer (see e.g. Bergland et al., 1995 and Alberini et al., 1997). One very simple approach, which is the one taken in Paper 3, is to adjust the estimates for income differentials, whereas a slightly more sophisticated approach includes taking income elasticity into consideration (the effect of this approach was briefly tested for mortality in Paper 3). A still more sophisticated approach is to transfer the WTP *function*. An appropriate adjustment of the function requires that the relevant site and population characteristics are represented as independent variables in the demand function, and that these can be estimated for the site to which the function is transferred (see also EC, 1995a, and Desvousges et al., 1992). Attempts to test different methods of benefit transfer have yielded ambiguous outcomes (see Alberini et al., 1997; Bergland et al., 1995; and Loomis, 1992).

In Aaheim et al. (1997) some of the ‘buts and maybes’ connected to the approach taken in Paper 3 are further discussed. The main conclusion is that WTP estimates from one country can be adjusted proportionately to income and applied for another country (as was done in Paper 3) only if certain conditions are fulfilled. These are, firstly, that the number of avoided cases per capita resulting from a given emission reduction is the same in the two countries, and secondly that the expenditure on health is proportional to income. It seems, at first sight, unreasonable that these conditions are fulfilled in our case. Theoretically, however, the first condition could be met if the exposure-response curves for the health effects were log-linear (as some studies in heavily polluted regions have indicated). Even though the pollution level and the number of affected people per capita probably are higher in Hungary that in the US, a certain amount of emission reduction in both countries could then result in the number of avoided cases per capita being close in the two cases.

WTP estimates for all end-point are not available in the literature and were therefore approximated by various methods (US-EPA, 1995 and Canadian Council of Ministers of the Environment, 1995, see Paper 3). One may have reservations concerning the robustness of some of the assumptions behind the approximations. It is, however, to the credit of the proposed estimates that they are transparent and easy to decompose in order to test the sensitivity of various assumptions to the final output of the analysis (see Section 6).

The sensitivity analysis given in Section 6 shows that various assumptions about the severity and duration of the chronic respiratory symptoms (CRS) have a large impact on the output of the health benefit assessment. The unit price for this end-point takes as a starting point the WTP for avoiding a rather severe case, which is found to be US$570000. This value represents the present value of the perceived welfare loss during the whole time span of the disease (see Paper 3), and thus is a so-called *incidence-based* unit price. Subsequently, the result from a study by Krupnick and Cropper (1992) was used to assess the WTP for a more average case of chronic bronchitis (hence, the term 'value of a statistical case', VSC, may be more appropriate, see EC, 1995b). Using a severity scale of 0-13 the WTP elasticity (V_e) with respect to severity was found to be 1.16 per 'severity point' (US-EPA, 1995). It is further assumed that the WTP for avoiding a case of severity *x* can be calculated as:

$$WTP_x = \{1 - I_p \cdot (13\text{-}x)\} \cdot WTP_{13}$$

where:
WTP_x is the WTP for avoiding a case with severity rating x, and
I_p is the growth rate of WTP per severity point, calculated as V_e times the step between each severity point (1/13)

The mean severity rating for the CRS cases evaluated by EPA was 6.5, hence the WTP becomes US$240000. This was adopted in the study described in Paper 3. The exposure-response function behind the estimated number of avoided cases of CRS referred to *differences in prevalence*. This implies that the prevalence of CRS in the population decreases gradually after the emission reduction has taken place, and that the total reduction in prevalence is not obtained until a certain number of people who became ill before the year of implementation (the 'lag' of cases), have recovered. To calculate the tentative benefit/cost ratio in the various sectors (see Figure 5 in Paper 3) we simply assumed that the reduced prevalence was obtained the first year after implementation, and thus avoided any discussion on the duration of CRS cases. As mentioned, the benefit attributed to CRS cases (see Table 10 in Paper 3) cannot be compared directly to the other end-points, as it is not the annual benefit. To make such a comparison, one has to make assumptions about the average duration of the cases in order to obtain an estimate of the *annual unit price*. However, not much information about duration was given in the original study. If 5 years duration is assumed and a 6% discount rate is used, the annual unit price becomes approximately US$54000 (US$8600 when adjusted for the wage ratio). The corresponding annual benefit attributed to reduced cases of CRS in children and adults, respectively, is 121 and 143 million US$ (instead of 541 and 637 million as given in Table 10 in Paper 3). This implies that the benefit of reduced prevalence of CRS would constitute 41% of a *total annual health benefit of 648 million US$.* As a comparison the benefit related to reduced mortality (infant mortality excluded) would then constitute 48% (see Section 6 for a sensitivity analysis).

For some of the estimates that we have applied it is assumed that the WTP may be derived from the cost of illness (COI - the sum of productivity loss and costs of medication, hospitalisation etc., see Section 5.1) by multiplying with a given WTP/COI-ratio. We agree that an uncertain adjustment is preferable to no adjustment (presupposing there are market imperfections). However, it must be admitted that the basis for stating the ratio in the US-EPA study seems rather week (the ratio was derived from a few empirical studies). On the other hand, the sensitivity analysis (see Section 6), showed that in our case various ratios had a moderate impact on the benefit output. Concerning the unit price for lung cancer cases the WTP/COI-ratio used for nonfatal cases was 1.5[6], and the weighed WTP for an average case was calculated as:

$$WTP_{weighted} = S \cdot WTP_{nonfatal} + (1\text{-}S) \cdot WTP_{fatal}$$
$$WTP_{nonfatal} = COI_{nonfatal} \cdot WTP/COI$$
$$WTP_{fatal} = VSL_{<65} \cdot D_{<65} + VSL_{>65} \cdot D_{>65}$$

where:
S: Survival rate
VSL: Value of a statistical life in the age group
D: Fraction of deaths in the given age group

As for CRS, the unit value for lung cancer is an incidence-based measure and refers to the present value of the cost of an average case. However, in this case the end-point in the calculation was annual new incidences of lung cancer cases, hence the physical end-point and the valuation start-point match. In the calculation of the tentative benefit/cost ratio given in Figure 5 in Paper 3, it was assumed that the number of avoided new cases is the same each year throughout the lifetime of the energy saving program. In the calculation of the present value of the benefit of emission reductions over the total lifetime of measures, the annual benefit of reduced new lung cancer cases is treated in the same way as the other end-points except CRS.

The procedures for estimating unit values for the other end-points are described in Paper 3. However, the unit values for premature mortality risks deserve special attention, because such estimates typically, and legitimately, are met with much more scepticism than are attempts to evaluate the welfare loss of health symptoms and even severe illness cases. They also highlight the differences between the *cost* of an effect and the social *value* of it. Pushed to its logical conclusion, a pure cost approach would mean that premature death of a person above the retiring age is socio-economically beneficial, a conclusion which would be of no relevance in a policy context.

The WTP for avoiding one premature death applied here is an expression of the estimated *value of a statistical life (VSL)*. It is crucial to note that such estimates are based on WTP of the individual for reducing his or her risk of premature death by a small amount, and do not intend to reflect the total value of a human life. For instance, if a study finds an average WTP of 300$ for an annual reduction in risk of death of 1/10000, this may be extrapolated to a per life basis by summing the individuals' WTP over 10000 people, giving a VSL estimate of 3 mill $ (US-EPA, 1995).

[6] This was proposed, without any clear argumentation, by the Canadian Council of Ministers of the Environment (1995). It may seem low, but on the other hand the COI is high for cancer cases as compared to many other diseases.

There are, however, indeed some unresolved problems and uncertainties in applying the VSL estimates in cost benefit analyses. The empirical evidence of WTP for reduced mortality risks is valid to a large extent only for accidental deaths in circumstances where individuals are voluntarily exposed to the risk factor. Premature deaths due to pollution exposure are related to illness rather than accidents and fall disproportionately on the elderly and those with an already compromised health. A substantial literature suggests that lost years of life may be a more appropriate measure in policy analysis (see e.g. Viscusi et al., 1997).

The various empirical and theoretical studies being the basis for the VSL proposed by the US-EPA (1995), and adopted here, use different models and are based on a variety of assumptions. For instance, the wage-risk studies estimate wage premiums associated with different levels of on-the-job risks, and often do not elucidate the importance of age to the VSL. The estimates also apply only to the population of risk exposed workers, who generally have lower incomes than average (Viscusi, 1993). Other types of studies using the so-called life-cycle consumption-saving model, generally indicate that the VSL peaks at the age around 40 (see US-EPA, 1995). This output is due to the model premise that utility is a function of consumption. Other types of studies assume a constant value per year of life, the future years are, however, discounted. This implies that the VSL decreases with age throughout a persons life, and how fast it declines depends on the discount rate. Finally, there also are studies indicating that the WTP for increasing the expected length of life by one year, conditional of having survived to the age of 75 years, increases with a person's age (the rate, however, seems to be low) (Johannesson and Johansson, 1996). Approaches taking into account the importance of age result in aggregated utility functions that are dependent on the age distribution in the population of interest.

In a statistical context most people would probably agree that the death of a younger person is a greater loss for the society than the death of an older person (see e.g. Cropper et al. (1997a) for an example of measurement of life saving preferences). This is reflected in the VSL used here, where the value of a statistical life under the age of 65 is somewhat higher than the value of a statistical life over the age of 65. The figures proposed by the US-EPA are within the range of results from a variety of studies, however it gives less weight to wage-risk studies. The weighting procedure that was applied results in the $VSL_{>65y}$ being about 75% of the $VSL_{<65y}$.

The empirical basis for VSL estimates are restricted to adults. Extrapolating the curves from existing studies to lower ages yields a VSL estimate going in either direction (see US-EPA, 1995). In this situation we decided to use the same estimate for all premature deaths under the age of 65.

An alternative to the cost-benefit approach to mortality risk management using VSL estimates, is the so-called health-health analysis (Chapman and Hariharan, 1996). The starting point for this approach is that rich people have a lower mortality risk than poor people, and that there is a break-even cut-off for investments in mortality reducing measures, above which the increased mortality risk due a reduced income level exceeds the reduced risk due the regulation. The empirical evidence of a non-linearity in the income-to-mortality linkage suggests that it is of crucial importance *who* bears the cost of the regulation. Chapman and Hariharan found that in the USA break-even cut-offs are roughly twice as large for the richest

20% of the population than they are for the poorest 20%. This means that if the rich pay the bill of a regulation the cost may be twice as high as if the poor have to pay it.

In the Hungarian case, where a great deal of the population is in a difficult economic situation, and the enhanced mortality rate probably is, at least partially, causally related to this, a health-health analysis approach might be of particular relevance.

5.3. Results from the Bottom-Up analysis – a tentative comparison of the benefits

The estimated annual health benefit from implementing the Energy saving program in urban Hungary was 648 mill. US$ (95% CI found by Monte Carlo simulation (see below) was 370-1168 mill. US$). For Budapest the estimated reduced damage to building materials amounts to 30-35 mill. US$. If linearly extrapolated to urban Hungary, it yields an annual benefit of 98-114 mill. US$. Finally, a rough estimate of the increased cereal crop production gave a range of 0.9-2.2 mill US$ (see Table 1).

Several attempts have been made to assess the benefits of reducing greenhouse gas emissions. In a review of studies trying to quantify external costs related to electricity generation, Krupnick and Burtraw (1996) found that all studies concluded that damage estimates related to climate change are too uncertain to be included with other estimates. The main focus of these studies was the air pollution impacts. In the summary of Working Group III of the Intergovernmental Panel of Climate Change (IPCC) it was concluded that practical application of traditional cost-benefit analysis to the problem of climate change is difficult because of the global, regional, and intergenerational nature of the problem, and that confidence in monetary estimates for important consequences is low. In the Chapter by Pearce et al. (1996) attempts to assess the benefits of reductions are summarised, and the marginal damage cost was found to be in the order of US$5-125 per ton carbon. The numbers are particularly sensitive to the choice of discount rate. Most of the estimates are in the range $5-20, but higher values were obtained if the discount rate is 2% or less. As noted by the authors, the models used to make these estimates are simplistic and are limited representations of the actual processes, but so far they represent the state of the art. Applying the interval $5-125 per ton carbon on the CO_2-reduction estimated to result from the energy saving program in Hungary, yields an annual benefit of 7-168 mill. US$ (Table 1).

The relative importance of the health benefit compared to other secondary benefits is in accordance with the majority of other studies. Naturally, it varies considerably depending on which effects have been included, and what are the important energy carriers in the cases, and a direct comparison is difficult (see overviews in Krupnick and Burtraw, 1996; Ekins, 1996; and Ayres and Walter, 1991). In most of the studies the health benefit constituted 70-90% of the quantifiable damages due to air pollution. In our study health effects constitute approximately 86% (using the best estimates). Considerably less effort was, however, put into the assessment of changes in crop loss and material damage related to implementation of the energy saving program, and the method for evaluating reduced material damage most likely only gives a lower bound for the full social benefit, as indicated in Section 5.1.

The heterogeneity of GHG reducing measures and the specific spatial distribution of the emission sources in question are likely to imply that diverging secondary benefits may be obtained from implementing GHG-abatement measures. This fact implies that a general 'best estimate' of the magnitude of secondary benefit obtained per ton of CO_2-reduction, which has been indicated in the economic literature (see e.g. Ekins, 1995), may be precluded.

Table 1: Estimated annual benefits from implementing the energy saving program in Hungary. Million US$.

Effect	Best estimate	Range
Health	648	370-1168
Building materials	105	60-150 [I]
Cereal crops	1.5	0.9-2.2
Climate change	25	7-168 [II]
Total	**780**	**438-1488**

[I] The uncertainties are subjectively estimated to be about 40%.
[II] Most estimates would be <30 mill. US$, see text.

5.4. Comparing the output of the top-down and bottom-up analyses

Generally, the B-U approach is considered to be appropriate when assessing small projects. The energy saving program assessed in the Hungarian case study amounts to almost 8% of the total energy consumption in Hungary. It is likely that implementing a program of this size would affect macroeconomic variables in a way that might result in a shift of the equilibrium in the economy. As we have argued, the special transient economic situation in Hungary would be rather difficult to represent adequately in a macro-economic model. In Aaheim et al. (1997) an attempt was, however, made to compare the various approaches by use of a simple and aggregated macroeconomic model based on traditional assumptions about market behaviour.

In a strictly economic sense, the value of a cleaner environment is defined only when the marginal willingness to pay equals the marginal cost for environmental improvements, that is, when the demand for an improvement equals the supply. This is where market equilibrium is established, and the market price is an expression for the value. Although an appropriate estimate of value is taken on the margin, i.e. the unit value related to any small additional change, one often sees in the literature that estimates of the total damage from x units of energy use are divided by the number of units to obtain the unit value of damage. This is only correct if all relationships in the impact pathway are linear (marginal and average damage are then equal). Unfortunately, these conditions rarely are satisfied, neither for the physical impact assessment nor for economic valuation functions (Kopp et al., 1997).

Figure 3 illustrates the principal differences between what is measured by the two B-U methodologies (WTP and damage cost, respectively) and a T-D approach. The thick MC-curve represents the marginal cost of energy saving measures, calculated as the cost per saved PJ of energy. Assume that the reduction in energy consumption from the Hungarian energy program equals x(1). The marginal cost of the program is p(1). The cost-benefit criterion of the program is that the marginal cost of the program is lower than the alternative price of energy. One might find that only a part of the project is socially beneficial. In a macroeconomic context, therefore, the whole program is sometimes regarded as marginal. Then, the test is whether the alternative price of energy exceeds the unit cost of the whole program.

The damage-cost approach is based on the assumption that the social economic benefit from energy saving should be subtracted from the marginal costs. This leads to a negative shift in the marginal cost curve, from MC to the marginal social cost curve, MSC. Hence, energy saving might be socially beneficial, even if the alternative price of energy is lower than p(1), that is, if the cost savings due to less energy use are smaller than the dark area under the MC curve in Figure 3. In the example displayed here, inclusion of environmental benefits turns the marginal social cost of the program negative, and from the viewpoint of climate policy, it would therefore be a so-called no-regret option. In such a case it is clearly beneficial to save more energy than x(1).

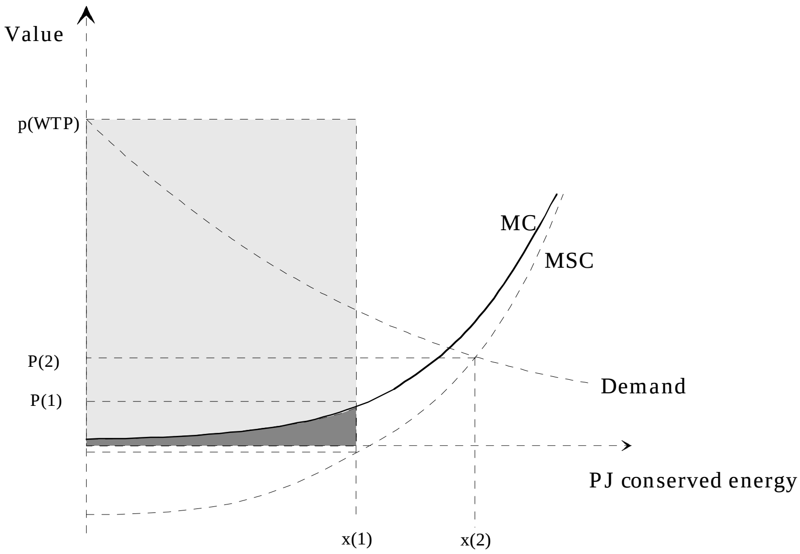

Figure 3: Alternative approaches to valuation of environmental change. MC: Marginal cost curve; MSC: Marginal social cost curve; x(1): energy saving program (assuming this is not optimal); p(1): marginal cost of energy saving program; p(WTP): willingness to pay per reduced PJ of energy use at x=0; x(2): optimal level of energy saving; p(2): unit price at equilibrium. (Figure modified from Aaheim et al., 1997).

Assume that the alternative price of energy is zero. By the willingness-to-pay approach, one attempts to examine the demand for environmental improvements, or for energy conservation. In a bottom-up analysis the estimated willingness to pay (p(WTP)) determines the point on the demand curve where no energy saving has taken place, i.e. at x = 0. Usually, it is required that p(WTP) should exceed p(1), if the measures are to be considered socially beneficial. This gives a net benefit (i.e. the benefit minus the cost) equal to the light grey area in Figure 3. This is not a perfect criterion, since the willingness to pay and the marginal cost refer to different states in terms of energy use, x(0) and x(1), respectively. Hence, for large changes, the willingness-to-pay approach may exaggerate the benefits, with an amount corresponding to the light grey 'triangular' area above the demand curve in Figure 3. If energy savings actually were carried out, the willingness to pay for less pollution would probably decrease. The curvature of this demand function was estimated in a simple top-down analysis, which means that the prices were endogenously determined within the model (Aaheim et al., 1997). p(2) in Figure 3 is the new equilibrium price where the marginal social costs equal the marginal willingness to pay, i.e. the point which would be established if energy saving measures were implemented and there were no market imperfections. The optimal amount of energy saving is indicated by x(2).

Both the damage-cost approach and the willingness-to-pay approach may give biased estimates if the measures are large enough to cause significant shifts in the market equilibrium. It is difficult to say which of the bottom-up methods is the best, but in some cases one approach may be better than the other. If the supply curve is 'flat' compared with the demand curve, a marginal cost estimate would approximate the equilibrium price better than the willingness-to-pay estimate. The willingness-to-pay approach applies well with a 'flat' demand curve. One could speculate that in Hungary the demand curve is likely to be relatively flat, at least if one assumes that the income level increases during the period of emission reductions and the income elasticity concerning environmental amenities is larger than one (see Paper 3 for a discussion of the latter assumption). The general decrease in WTP for a given additional improvement would then be counteracted by an increase in WTP related to the income level becoming higher. The data that were available on implementation costs of the program indicated that the cost curve becomes very steep beyond an energy saving quantity of 40-50 PJ. However, these data were rather incomplete, and the shape of the curve therefore uncertain.

For practical purposes WTP might be considered an upper bound, whereas COI indicates a lower bound for the benefit. A full analysis of the energy program, however, requires a macroeconomic model, which includes a specification of the energy saving program and relations between economic activities and environmental effects of energy saving. This is particularly important in cases where the supply and demand curves are steep, and small shifts may lead to significant changes in relative prices.

In the study by Aaheim et al. (1997) the two B-U approaches, damage cost and willingness-to-pay, gave very large differences in the estimated annual benefit of the energy saving program, 75 mill. US$ and 648 mill. US$ respectively[7]. When WTP or damage cost estimates are used as the indicator of value in a bottom-up analysis, these refer to the initial state, before the emission reductions were implemented, i.e. in a state of market failure (i.e. to the left of the market cross in Figure 3). If the respondents behind a WTP estimate were 'ideal', i.e. perfectly disposing full information, and the damage cost estimates likewise were including all aspects of the damage done, one could maintain that market failure is *the only reason* why the two approaches, WTP and damage cost, diverge. In practice, however, neither of them is likely to be perfectly comprehensive and they do not include the same aspects. WTP covers broader considerations than a damage cost estimate, as for instance welfare losses due to suffering, pain, and anxiety (see Navrud and Pruckner, 1997). Symptomatically, much of the difference between the B-U estimates in the study by Aaheim et al. is due to the diverging

[7] The model calculations rendered in the Working Paper by Aaheim et al. (1997), unfortunately were based on an erroneous input estimate of the WTP for reducing chronic respiratory diseases. The figures given in the following are results from a new model run, where this was corrected, hence they may differ from the figures given in Aaheim et al.

estimates for chronic diseases and premature mortality, end-points where welfare losses are likely to be more prominent than for many other end-points. The extent to which WTP exceeds COI, however, is state dependent, and it seems unreasonable to apply a *general* WTP/COI-ratio in cost-benefit analyses, because it will decline as the state of equilibrium is approached.

The implementation cost of the program was indicated to be around 66 mill. US$ annually (see Aaheim et al., 1997), and both of the two B-U approaches thus result in the recommendation that the total program should be implemented, even when the alternative value of energy is set to zero. In the T-D analysis the estimated willingness-to-pay for a lower risk of health impact given above was used to estimate the demand curve in the model. Also, it was assumed that the cost curve for the energy saving measures could be prolonged beyond the amount of energy saving that constitutes the program. The estimated market equilibrium point implies a recommendation that energy saving measures corresponding to 64.2 PJ, x(2) in Figure 3, should be implemented (i.e. very close to the original program of 63.7 PJ). The defined unit value of *environmental quality*, i.e. the price of energy saving, p(2) in Figure 3, was about 6.9 mill. US$/PJ. The corresponding total abatement cost, i.e. the integral under the cost curve in Figure 3, was 41.2 mill. US$. These estimates are, however, very sensitive to the various assumptions made. When the WTP was reduced to ½, the model reduced the equilibrium energy saving quantity to 59.2 PJ, i.e. slightly less than the original program. Due to the steep cost curve the corresponding total abatement cost then fell drastically to 16.9 mill. US$, and p(2) became 3.1 mill. US$/PJ. Moreover, when the cost curve for the measures was assumed to be somewhat more gently sloping (i.e. taking off at an earlier stage and being less steep at higher abatement levels), the equilibrium energy saving quantity became 71.3 PJ. Total abatement cost then was estimated to be 73.9 mill. US$, and p(2) became 6.1 mill. US$/PJ.

6. Sensitivity analysis

The purpose of a sensitivity analysis is to evaluate the robustness of the outcome of calculations. If the outcome is very sensitive to parameters and variables which are based on limited or uncertain data, the confidence in the results will be low. If there, on the contrary, is a good basis for determining the parameters to which the output is more sensitive, there should be a high degree of confidence in the estimates. In Table 2 the sensitivity to input parameters in the physical damage assessment is shown. The sensitivity to each tested parameter is indicated by the relative change in the output (both in terms of the physical damage unit for each specific end-point and the aggregated monetary benefit for all end-points) produced by a 10% reduction in the input parameters.

To some extent the reliability of the parameters is indicated by their standard deviation (SD). In Table 2 the column 'sensitivity rating' gives the product of the sensitivity (calculated as the percentage reduction in the benefit estimate per percent point shift in the parameter), and the percentage points that one SD constitute of the central estimate of the parameter in question. The obtained figures give a quantitative indication of the uncertainty in the total benefit estimate connected to a 10% shift in each input parameter[8]. A general rule of thumb might be that if this figure exceeds 5 there is a worrisome uncertainty connected to the parameter. By normalising these figures a 'sensitivity rating index' (SRI_{10}) is obtained. Since SRI_{10} represents the impact of a 10% change in a parameter on the output weighted by the degree of uncertainty in the parameter, it gives an indication of how a parameter contributes to reducing the confidence of the analysis *relative* to the other parameters.

The robustness of the analysis as a whole, however, remains a matter of judgement, taking all aspects into consideration. For instance, there may be uncertainties in the parameters that are not necessarily reflected in the confidence interval. And as discussed in the next Section, there are model uncertainties.

As seen from Table 2 a shift in the two regression coefficients in the NO_2-to-PM_{10} conversion function (see Aunan et al., 1997) gives the largest shift in the benefit estimate. An important reason for this is that this shift affects the exposure assessment and thus affects all end-points. The most conspicuous shift in the physical damage estimates arises from reducing the regression coefficient, β, in the exposure-response function for chronic respiratory symptoms in children (CRS-Ch). The uncertainty in this parameter is by far the largest, and SRI_{10} indicates that nearly 40% of the 'confidence reducing capacity' is due to this end-point (monetary valuations taken as a premise). In addition to being uncertain, the regression coefficient is high compared to the coefficients for other end-points. Also, the unit value is rather high, hence a shift in the physical output is accompanied by a considerable shift in the benefit outcome. As discussed in Paper 2, it is very difficult to establish exposure-response functions for chronic diseases, *i.a.* due to the time lag in response and the problem with finding an appropriate averaging time for the exposure assessment. The function proposed in Paper 2 and used in the Hungary study is based on one single epidemiological study, and the uncertainty interval is wide, hence one could legitimately maintain that the basis seems weak.

[8] If all relationships in the model had been linear, these figures would correspond to the percentage change in the benefit output obtained from a one SD shift in the parameter. In our case they deviated slightly due to non-linearity, especially for CRS-Ch.

On the other hand, the study is a large cross-sectional study in six cities (in the USA), where respiratory illness and symptoms in the studied cohort of school children had been thoroughly followed and re-examined over a period spanning several years. Thus, the study in itself is more comprehensive than most other epidemiological studies. In Table 2 the impact of using a reduced regression coefficient indicated by the authors (accounting for possible recall problems; β becoming 0.018, see Paper 2) is shown to halve the physical response estimate.

Several assumptions were made to obtain unit value estimates for the health end-points. Table 3 shows the result of a sensitivity analysis of these assumptions. The assumed severity and duration of the chronic respiratory symptoms are the most decisive input parameters. Concerning the severity rating this is due to the calculation procedure employing a rather high elasticity of WTP with respect to severity (see Section 5.2). The important question is the robustness in the assumption about severity made first in the EPA report, which seems reasonable, and secondly in the adaptation of the value in our case study. As discussed in Paper 3 it seems unlikely that the degree of severity of the cases should differ substantially, and a severity below 5 is very unlikely. Now, if the symptoms estimated by the exposure-response function in the case study in Hungary were less severe than those given an unit value in the EPA-study, the baseline frequency (p_0) used in our calculation would possibly also have been miss-classified (severe cases are less frequent than the lighter), which would *amplify* a possible error in the benefit estimate. As indicated in Table 2 the sensitivity of the parameter p_0 in itself is, however, not very high for neither of the end-points CRS-adults nor CRS-children.

The assumed duration of CRS cases has a large impact on the benefit estimate. If the average duration of a CRS case given a unit price in US-EPA (1995) is 10 years or 15 years, respectively, the annual unit price drops to 57% and 43% of the price obtained if 5 years duration is assumed. Whereas CRS cases constitute 41% of the total annual health benefit assuming 5 years duration, this figure drops to 28% and 23%, respectively, if 10 years or 15 years are used in the calculations.

Table 2. Sensitivity of a 10% shift in the input parameters related to the central physical output estimate for each end-point, and related to the central estimate of the total health benefit of the energy saving program (ad. sensitivity rating, see text).

Parameter[I]	**% of physical estimate**	**% of total benefit estimate**	**SD/Central parameter**	**Sensitivity rating**	**SRI_{10}**
CRS-Ch, β	80.7	96.4	0.48	17.32	0.382
CRS-Ch, β=0.018[II]	51.0	91.1			
NO_2-PM_{10}-conversion[III]		87.8	0.09[IV]	10.90	0.241
Mortality>65y, β	91.7	96.0	0.11	4.47	0.099
CRS-Ad, p_0	86.7	98.0	0.20	3.95	0.087
CRS-Ad, β	86.3	97.0	0.10	3.16	0.070
Infant mortality, β	85.7	99.4	0.28	1.54	0.034
ARS-Ch, β	87.0	99.6	0.37	1.41	0.031
Mortality<65y, β	89.9	99.3	0.10	0.71	0.016
Lung cancer, β	90.1	99.8	0.31	0.67	0.015
ARS-Ad, β	88.4	99.8	0.24	0.56	0.012
CRS-Ch, p_0	91.8	98.5	0.02	0.26	0.006
Asthma days, β	90.0	100.0	0.50	0.23	0.005
ARS-Ad, p_0	90.4	99.8	0.04	0.07	0.002
ARS-Ch, p_0	91.4	99.8	0.02	0.05	0.001
Infant mortality, p_0	100.0	100.0	0.20	0.00	0.000

[I] See Paper 3 for definition of the end-points and explanation of the abbreviations.
[II] *Not* a 10% shift, see text.
[III] Two parameters.
[IV] The figure is the average quotient for all cities for ($PM_{10\ central}$ – $PM_{10\ low}$)/$PM_{10\ central}$, where PM_{10} is calculated using, respectively, central or low estimates of the conversion factors (for NO_2-PM_{10}).

Table 3. Sensitivity of a shift in the input parameters related to the central estimates of the annualised unit value for each end-point, and related to the central estimate of the monetary benefit of the energy saving program.

Economic valuation parameter	% of unit value estimate	% of total benefit estimate
ARS-HA-ratio: 0.0025[I]		98.3
ARS-RAD-ratio: 0.05[II]		99.6
Severity of CRS: 4[III]	47	78.3
Severity of CRS: 5[III]	68	87.0
Severity of CRS: 6[III]	89	95.7
Duration of CRS: 10 years [IV]	57	82.6
Duration of CRS: 15 years [IV]	43	76.9
Lung cancer survival ratio: 0,1[V]	111	100.2
Lung cancer deaths ratio> 65y: 0.5[VI]	106	100.1
WTP/COI-ratio cancer: 3.5[VII]	102	100.1
WTP/COI-ratio HA: 3[VIII]	150	101.7

[I] Share of hospital admissions (HA) of total acute respiratory symptom days (ARS). Instead of 0.005.
[II] Share of restricted activity days (RAD) of total acute respiratory symptom days (ARS). Instead of 0.1.
[III] Instead of 6.5, see Section 5.2.
[IV] Instead of 5 years (6% discount rate).
[V] Instead of 0.2.
[VI] Instead of 0.7.
[VII] Instead of 1.5.
[VIII] Instead of 2.

7. Uncertainty estimates

Policy makers have an inclination to want the output of consequence analyses to be single figures that are easily perceived in the context of decision making. However, point estimates, although simple, accessible and readily accepted by regulators, provide no information about the uncertainties embedded. A consensus is emerging that risk analysts should put more emphasis on quantifying the degree of scientific uncertainty in their estimates. The rapidly increasing power of desktop computers in recent years has also made analyses of the full probability distribution of a given outcome more readily computable (Evans et al, 1994 and Burmaster and Stackelberg, 1991).

Various terms for the procedure of calculating the probability distribution instead of (or in addition to) a point estimate with an uncertainty interval, are stochastic simulation, Monte Carlo simulation, and the probabilistic or distributional approach. Some advantages of the approach are that one may avoid disputes over best point estimates, the risk estimates are associated with a quantitative measure of uncertainty, and it allows for quantitative evaluation of a possible conservatism in point estimates. One must, however, be aware of possible interdependencies between variables (Finley and Paustenbach, 1994) and of error sources not included.

The need for explicitly representing uncertainties is closely related to the various decision rules discussed above in Section 2. The inevitable trade-offs that are made between various objectives will become more transparent when probability distributions are available, and one may say that Monte Carlo analysis separates risk assessment from the risk management (Burmaster and Stackelberg, 1991). For instance, for policy makers to be able to consider a *de minimis* standard or the *maximin* criterion in practise, it is a prerequisite that the analysis reveals the probabilities of outcomes in the lower and upper bounds. Hence, it is a prerequisite that probability distributions actually can be attributed to the various input parameters and variables. Often objective knowledge is lacking, however the belief of and agreement among experts may be seen as a legitimate measure of likelihood (the so-called Bayesian concept of subjective probability (see e.g. Evans et al., 1994).

Three main types of uncertainty may be identified in environmental risk analyses (see e.g. Trønnes and Heiberg, 1989): 1) The processes involved are inherently stochastic in nature, or at least so complex that it is infeasible to build precise deterministic models; 2) reliable mathematical models cannot be formulated due to the process in question being incompletely understood; and 3) plausible functional relationships for the process have been established, but there is some uncertainty connected to the input parameters. In health risk analyses usually the two last types represent a problem, and it is useful to distinguish them as model and parameter uncertainty, respectively. Monte Carlo simulation is particularly useful in treating parameter uncertainty, although it may also be used in characterising model uncertainty (see Trønnes and Heiberg, 1989, and references therein).

The calculation of the health benefit of energy saving in the Hungarian case study presented in Paper 3 relies on the applications of several sub-models, *i.a.* linear relationships were assumed between emission, concentration level, and population exposure. Moreover, the functional form of each exposure-response function applied represents uncertainty, and the degree of uncertainty may vary substantially between the end-points. Probably, the uncertainties connected to the use of exposure-response functions by far surpass the uncertainties connected to the assessment of reduced exposure.

In the following, Monte Carlo simulation is used to assess the aggregated uncertainty in the total health benefit estimate arising from uncertainties in some important input parameters and variables. Figure 4 shows the resulting overall probability distribution. 10000 draws were made for: The factor used in estimating PM_{10}-levels from NO_2-levels, the conversion factor for $PM_{2.5}/PM_{10}$- and PM_{10}/TSP-ratios, the regression coefficients in all exposure-response functions, the baseline frequencies of end-points (p_0) when this was needed, the ratio of asthmatics in the adult population, the share of acute respiratory symptom days resulting in hospitalisation, and the monetary unit values found in western studies (taking into consideration the various input parameters in these). In order to reduce the number of iterations a population-weighted NO_2-level was estimated based on the data from all cities and draws were made from a log-normal distribution.

The draws for each parameter were made from Normal distributions (using the standard deviation), except for those parameters for which we only had estimated a likely range, as the hospitalisation rate and the share of asthmatics. For these the draws were made from a uniform distribution within the range regarded as plausible. Principally, none of the parameters that were drawn in the stochastic simulation are interdependent. However, since several of them are derived from western studies, one could maintain that they are interdependent in the sense that they are 'westerly biased'. This, on the other hand, relates to the model uncertainty more than to the parameter uncertainties. The need for distinguishing model from parameter uncertainty is also the rationale for not making draws for the valuation multiplier (which was 0.16, see Paper 3). There is practically no uncertainty in the factor itself, however, the whole procedure of using the valuation multiplier entails numerous methodological problems. As noted by Kopp et al. (1997) the methods for establishing error bounds and central estimates still are ad hoc and vary across benefit-transfer studies.

The histograms resulting from Monte Carlo simulations are often highly non-Gaussian in shape. In our case the exposure assessment has a log-normal distribution, as showed in Section 4.1. Also the exposure-response function for several end-points gives a skewed distribution. However, even if all inputs had Gaussian distributions, the final result is likely to be non-Gaussian, since the input variables enter the formulae by multiplication, division and subsequent summation (Thompson et al., 1992).

The utility theory, as mentioned in Section 2, advocates the rationality in using the *expected value* as the decision criterion, irrespective of distribution. The expected value (i.e. arithmetic mean) of the distribution given in Figure 4 is 686 mill. US$[9]. The geometric mean is 658 mill. US$, whereas the value of the total health benefit obtained by using central estimates of all parameters in the deterministic model was 648 mill. US$. Geometric mean values are often used for representing the central tendency of data that have a log-normal distribution. However, they are biased low due to high values being downplayed through the logarithmic transformation (see e.g. Parkhurst, 1998). Provided that the probability distributions attributed to the parameters in the Monte Carlo simulation are reasonably well founded, the arithmetic

[9] 5 years duration was assumed for CRS cases, see Section 5.2.

mean would represent a better estimate of the central tendency than both the geometric mean obtained from the simulation and the central estimate obtained from the deterministic model.

The Monte Carlo simulation showed that for instance there is a 50% probability that the benefit is between 550 and 800 mill. US$. There is also a 50% probability that it is between 600 and 900 mill. US$. The outcome of the simulation may be used to estimate a 95%CI for the total benefit, taking as a starting point the standard deviation of the log-transformed output of the simulation. If the mean value (whose anti-log is the geometric mean of the distribution) is used directly in the calculation of a 95%CI, the range is 370-1168 mill. US$. If instead the arithmetic mean is used (log-transformed) the interval shifts somewhat and gets broader, 386-1219 mill. US$. Following the same argumentation as above the latter should perhaps be preferred, because the first is biased low.

The low and high estimates for the economic benefit connected to each end-point given in Table 10 in Paper 3 are the results of using all the low and high input parameters, respectively. By means of the Monte Carlo simulation 95% CIs for these figures could be estimated, see **Figure 5**. The central estimates given in the Figure are those obtained by using central estimates of all parameters in the deterministic model, and the CIs are calculated directly from mean values of the log-transformed distributions in the simulation. Table 4 shows that the central estimates from the deterministic model generally lie between the geometric mean and the arithmetic mean obtained in the Monte Carlo simulation[10].

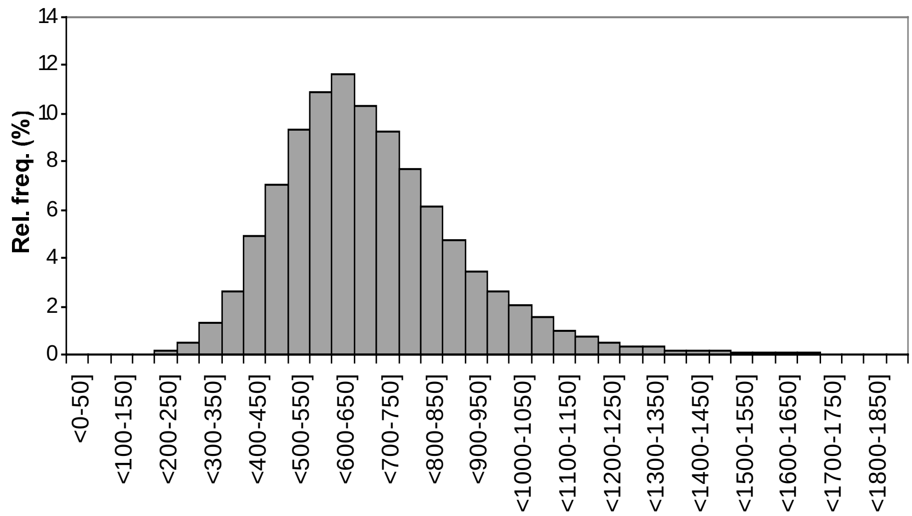

Figure 4: Probability distribution for the total health benefit of implementing the energy saving program in Hungary. 10000 trials.

[10] The sum of individual geometric mean values obtained from stochastic simulation is not the same as the geometric mean of the total distribution shown in **Figure 4**. This is due to the differing individual distributions for the end-points.

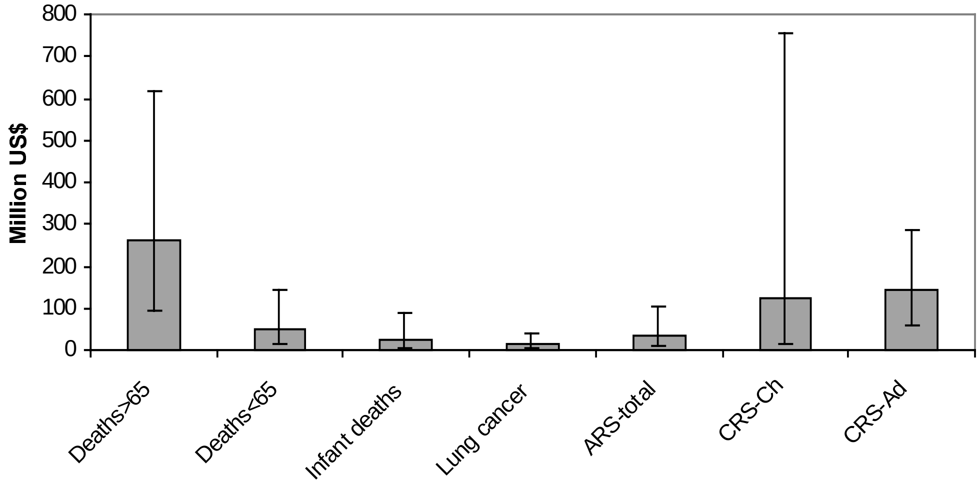

Figure 5: Central estimates and 95%CI for health end-points.

Table 4. Geometric mean and arithmetic mean of health benefits obtained by Monte Carlo simulation, and central estimates of health benefits obtained by running the deterministic model.

	Million US$		
End-point	**Geometric mean (M-C-simulation)**	**Central estimates (deterministic model)**	**Arithmetic mean (M-C-simulation)**
Deaths>65y	242.5	261.2	264.6
Deaths<65y	46.6	51.1	52.0
Infant deaths	20.6	24.8	25.4
Lung cancer	11.1	12.8	13.0
ARS-total	29.5	34.3	34.9
CRS-Ch	103.0	121.3	153.6
CRS-Ad	133.0	142.8	142.7

8. Conclusions

The basis for performing quantitative analyses of the physical impacts of air pollution has grown steadily more firm during the last decades, and the use of exposure-response functions seems warranted for a number of effects. However, for some important effects there are still large uncertainties and doubts connected to whether present knowledge justifies application in policy analyses. Concerning health effects this especially applies to various chronic diseases that are associated with air pollution exposure. The work described here indicates that air pollution may have a large impact on the prevalence of chronic health diseases and symptoms, and that the entailed economic impacts may be very large. The uncertainties in the estimates are, however, much larger for these effects than for many other health end-points. Concerning agricultural crop production there also still are very large uncertainties connected to use of the dose-response approach. For building materials there is a good basis for estimating the damage by use of dose-response functions.

Estimates of the economic benefit of less pollution may differ significantly depending on the choice of method. Bottom-up approaches, using damage costs or willingness to pay assessments and top-down approaches are therefore often considered to be alternatives. An important conclusion from the case study in Hungary is that the approaches support each other, and that both are needed in order to do an appropriate assessment of the benefits. The damage cost approach provides vital information about the economic consequences of reduced emissions to air. The willingness to pay approach is used to include the demand for improved environmental quality. Willingness to pay is likely to differ substantially between populations and in our case study in Hungary relevant studies of willingness to pay were not available. Finally, in order to find the cost-efficient level of emission reductions a top-down analysis is needed. For those environmental problems that have a global, regional, and intergenerational nature, cost-benefit analyses, however, have some clear limitations.

The fact that the various methods may give widely different results calls for prudence whenever using the outcome of such analyses in a policy context. Whereas the estimates of physical damage may be very uncertain they are, at least in principle, value free. When the physical damage is transformed into economic damage in terms of its effect on labour supply and productivity, the uncertainties accumulate additionally, but the estimates are still in principle value free. At the moment one attempts to include the welfare effect, however, the fundamental methodological problems tower and it may be as informative to do sensitivity analyses as to try to estimate an adequate uncertainty range.

Although usually regarded as more straightforward than assessing the benefits, our experience is that estimating the marginal cost curve for measures to reduce energy consumption also can be rather difficult. This was due to the scarcity of data on implementation costs and data needed to estimate the reduced damage costs, and to the situation in Hungary, being a country in economic transition.

Concerning health effects that span several years, it is important to recognise the difference between measures of prevalence versus incidence rates. A lack of accordance between the physical end-points and the economic valuation start-points may represent a problem in studies like ours. Concerning chronic respiratory diseases and symptoms more long-term longitudinal studies on incidence rates would be beneficial and should be encouraged.

Whatever called, integrated assessment or cost-benefit analysis, it seems that a credible analysis should involve the results of a multitude of disciplines. In addition to the challenge of understanding the relationships of relevance within ones own discipline, such work requires 'compatibility' in the links between different parts of the analysis. It enhances the quality of the analysis if information on the probability distribution of the outcome is provided. Due to the large sources of uncertainty the value of performing analyses of this kind may seem limited. Maybe the process in itself is more important than the results; *"all models are wrong, but some models are useful"*, as noted by Shlyakter (1995). Obtaining a picture of the variety of effects, how different factors influence others, and performing sensitivity analyses for important parameters undoubtedly contribute to more consistent decisions within the field of environmental protection. Moreover, it is a question what are the alternatives to such analyses.

Concerning the energy saving program in Hungary, the recommendation that it should be implemented seems clear from our study, even *without* taking into account its contribution to reduced greenhouse gas emissions, which after all was the original purpose of the program. Our study demonstrates the importance of considering 'secondary benefits' in efforts to reduce greenhouse gas emissions, at least whenever such benefits are likely to be prominent. And they certainly are, in the industrialised countries, as well as in large areas in the developing world, in the foreseeable future.

Acknowledgements

The present study has been carried out in the period from May 1993 through February 1998 at CICERO. Apart from CICERO, the Norwegian Research Council has provided financial support through their 'Scholarship program for female environmental researchers'.

The study was initiated by Professor Hans Martin Seip, who has also been my supervisor. I am very grateful for having the opportunity to work together with him. He has been a most inspiring supervisor, and has a remarkable insight, in breadth as well as in depth, into the field of environmental science.

It has been a great pleasure to work together with Asbjørn Aaheim, the economist in the project. He has had an apparently boundless patience with me, a natural scientist trying to understand some basic concepts of economics. The interdisciplinary approach of our study I think has been very fruitful and is highly recommended to those of you working with the issues of environmental protection.

I am indebted to our Hungarian co-workers, especially Dr. Tihamér Tajthy and Dr. György Pátzay at the Technical University of Budapest. Tihamér Tajthy and his wife Edith Tajthy provided much of the data needed to perform the analysis of possible effects of reduced air pollution in Hungary, and I am very grateful for having had the opportunity to get to know these knowledgeable, helpful and hospitable people.

I would like to thank Dr. Per E. Schwarze at the Norwegian National Institute of Public Health for useful guidance and comments concerning the part of my thesis dealing with human health effects of air pollution.

Also, I would like to thank the rest of my colleagues at CICERO, taking the chance to accentuate Jan S. Fuglestvedt and Tora Skodvin, for providing a very pleasant, friendly, and stimulating work environment.

Finally, thank you Terje for being a great companion in life, encouraging and cheerful, and for always being there for our three kids, one of whom was born during this period. And thanks to you Kasper, Bendik and Ylva for being the best kids a mother could ever want.

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CICERO was established by the Norwegian government in April 1990 as a non-profit organization associated with the University of Oslo.

The research concentrates on:

- International negotiations on climate agreements. The themes of the negotiations are distribution of costs and benefits, information and institutions.

- Global climate and regional environment effects in developing and industrialized countries. Integrated assessments include sustainable energy use and production, and optimal environmental and resource management.

- Indirect effects of emissions and feedback mechanisms in the climate system as a result of chemical processes in the atmosphere.

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