Report 1997:1

Health and Environmental Benefits from the Implementation of an Energy Saving Program in Hungary

by

Kristin Aunan, György Pátzay*,
Hans Asbjørn Aaheim and Hans Martin Seip

January 1997

* Technical University of Budapest, Dept. of Chem. Tech.,
H-1521 Budapest, Budafoki u.8 F.II.7, Hungary

ISSN: 0804-4562

INTRODUCTION

Many of the countries in Central and Eastern Europe that previously had a communist regime have been striving with severe economic recessions. In Hungary economic stagnation is an important reason why the fossil fuel consumption, and thereby emissions of air pollutants and greenhouse gases, was considerably reduced in the period from ca. 1980 until the mid 1990s. However, in spite of large overall reductions air pollution still causes problems to human health and environment, and the emissions are expected to increase towards the turn of the century. This fact, together with international obligations concerning regional and global environmental issues, and Hungary's rapprochement to the European Union, necessitates enforcement of a comprehensive environmental policy in Hungary.

The demand for economic growth, and the problems in achieving it, is likely to sharpen the conflict between different goals, hence cost-effectiveness in environmental management becomes essential, maybe even more in Hungary than in established Western market economies. In the process of establishing a long-term environmental policy in Hungary, it may be helpful to relate experiences from other countries to the Hungarian case. This paper is a part of a study which aims at contributing to this process (see Seip et al., 1995). The main objective in the following is to make a rough assessment of some potential benefits that could be obtained from reducing emissions of air pollutants in the different economic sectors. Such estimates could help to set priorities in air quality management. The assessment takes as a starting point abatement measures within the Action Program for Energy Conservation developed by Hungarian authorities during the last years. We have made estimates for effects on health, building materials and cereal crops.

INTEGRATED ASSESSMENT

Basically two approaches may be used to find cost-effective abatement strategies against pollution damages (Aaheim, 1994; Aunan et al., 1995). In the "top-down approach" (T-D) the assessment is done by the use of macroeconomic models, which are particularly suitable for analysing the impact of indirect measures, such as taxes, on main macroeconomic variables. From the predicted changes in economic activity the emission reductions are deduced, and the benefits from these reductions may be fed back into the macroeconomic variables. In the "bottom-up approach" (B-U) specific abatement measures considered appropriate for solving a problem are explored in detail. Their potentials for reducing adverse exposure of recipients (people, crops, forests, materials etc.) and thereby damage, are estimated. Assessments of the values of the costs and benefits are then made according to observed or estimated market prices. To a large extent monetization of environmental and health qualities depends on subjective valuation and various methods have been applied for this purpose, all of them have weaknesses and problems (OECD, 1989; Navrud, 1994; Wenstøp et al., 1994). The social net benefit provides the basis for a ranking of measures.

The T-D and the B-U approaches both have major weaknesses: While T-D analyses tend to oversimplify for instance the biogeochemical relations, the B-U analyses tend to oversimplify, or simply leave out, macroeconomic relations and consequences.

Our approach is the B-U, and focuses the *damage assessment*, i.e. the relations between emission sources, concentration levels, exposure and effects on health, vegetation, materials and climate. This approach has advantages in explicit valuation of environmental amenities, and provides means to assess environmental values not directly related to damage costs. Besides, the special transient economic situation in a post-communist country like Hungary at present would be difficult to represent adequately in a macroeconomic model. It is, however, necessary to analyse the political, institutional and socio-economic environment within which abatement strategies will have to be chosen, in order to make the analysis realistic. These factors are not dealt with in this paper, since we take as our starting point a set of measures already proposed in an Hungarian energy saving program (see Section 4).

EMISSION TRENDS IN HUNGARY

In Hungary really critically polluted areas are less frequent compared to other Central and Eastern European countries (CEE), and only a few "hot spots" can be named, e.g. the Sajo valley, the Transdanubian industrial districts and Budapest. Besides, the difference between these areas and other parts of the country with more typical levels of pollution is less pronounced than in other countries in this part of Europe (REC, 1994). However, since the most polluted areas are also the most densely populated, many people are exposed to adverse concentrations, 44% of the population according to the Hungarian Ministry of Environment and Regional Policy (1992).

General economic recession, increased use of nuclear energy, some specific abatement measures, and structural changes in the economy, have caused large changes in the emissions of air pollutants in Hungary in recent years. In many ways the changes display the same pattern as in Western Europe a couple of decades ago, industry becoming less important as a pollution source and the transportation sector becoming more important. A predominant feature is that the private transportation does not seem to be prevented from growing even by deep economic recession. In Hungary in the period 1985-1992 the number of passenger cars increased by 43%, the gasoline consumption increased by 17%, while GDP decreased by 13%. Changes in total fossil fuel consumption in the different sectors varied considerably: The largest decreases were in agriculture (59%) and industry (39%).

In addition to these overall large reductions, there has been a tendency towards more use of gas, see Figure 1. In 1980 the solid fuel accounted for 35%, compared to gas 29%, whereas the corresponding figures in 1994 were 24% and 41%. This has contributed to cleaner air in many areas. Fossil fuel consumption peaked around 1980, but the total energy consumption (including nuclear energy[1]) was at its highest somewhat later, around 1985, followed by a marked reduction the following years.

[1] Nuclear power was introduced in 1983. In 1985 the share of nuclear fuel in the total fuel consumption (in PJ) of public power plants was 20.5%, almost doubling to 38.4% in 1995.

As a result of the above mentioned changes the emissions of greenhouse gases (GHGs) and air pollutants have been reduced. In the period 1985-1992 the reductions were approximately: CO_2: 26%, SO_2: 40%, TSP (total suspended particulates): 60%, NO_x: 28%, CO: 21%, nmVOC (non-methane volatile organic compounds): 45%[2]. The dominant GHG from fossil fuel use is CO_2, both in tons and estimated as future integrated radiative forcing. The uncertainties in the emission data given are estimated to ±15% (see also Tajthy, 1993; Tajthy et al., 1990; and Seip et al., 1995).

Heat and electricity generation (public power plants) are the most important sources of SO_2 (53% of total) due to the fact that coal and lignite with relatively high sulphur content still are important fuel types, and the general lack of desulphurization equipment. Households are the dominant source of particulates (42%) due to the widespread use of coal and coke briquettes. The transportation sector is dominant as regards NO_x (54%), and plays an increasing role in the continuing deterioration of urban air quality. Traffic has become the fastest growing urban air pollution source in the past 15 years. Two-stroke engines in the vehicle park are relatively frequent, and the phasing out is likely to be retarded due to economic recession, unless specific action is taken. Concerning passenger cars in general, in 1992 35% were older than 12 years (Hungarian Central Statistical Office, 1992), which implies that they have high specific emissions of many components. A reduction in the content of lead in the gasoline (from ca. 0.35g/l before 1992 to ca. 0.15 g/l from 1992), and the introduction of unleaded gasoline (8% in 1992), however, gave a 60% reduction in the emission of lead from this sector in 1992 as compared to 1985.

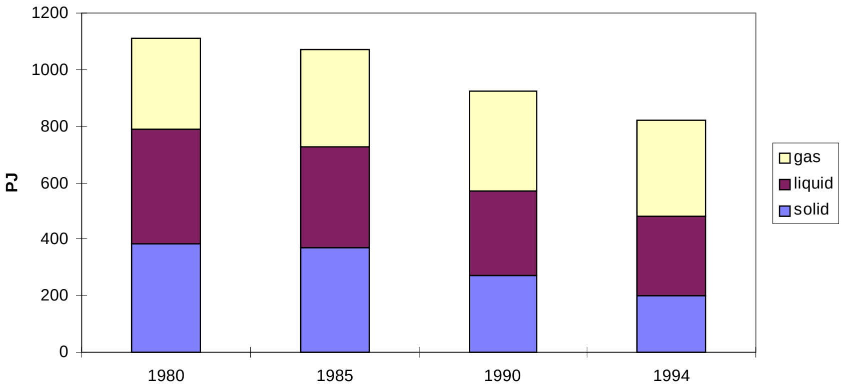

Figure 1. Fossil fuel consumption in Hungary, 1980 - 1994.

[2] The emission figures used in this paper are somewhat lower than the figures reported to EMEP, partly because our figures apply mainly to emissions connected with fossil fuel combustion. The difference is particularly large for nmVOC. For instance evaporation from solvent use, paints, fuel distribution etc. is not included. In the work on verification of the emission data at EMEP/MSC-W, Berge et al. (1995) notes that the CORINAIR90 inventory reported 28% lower Hungarian VOC-emissions than what was reported to the UN-ECE and used in the EMEP-model. The figures used in our study is 64% lower than assumed in the EMEP-model (data for 1990).

AIR POLLUTION ABATEMENT MEASURES IN HUNGARY

An effective policy towards air pollutants and greenhouse gases is largely determined by the degree of integration of environmental concerns into main policy areas in a country, especially concerning energy, transport and industry policies. In Hungary an integration has been hampered by the economic recession, which is likely to have resulted in a decreased concern about environmental issues.

The Hungarian energy system has been characterised by a dependence on energy imports on the supply side and high energy intensities on the demand side. The high import dependency has entailed limitations and vulnerability due to the existing transportation facilities (e.g. pipelines) and the lack of political stability of exporting countries, mainly former USSR countries (Kovacsics et al., 1994). An important objective of Hungarian energy policy is therefore to obtain a higher reliance on domestic energy sources on the one hand and to diversify the import on the other hand, mainly by connecting to the European networks. Unless the potentials for renewable energy are exploited, an increased dependence on domestic energy sources implies that the energy policy is increasingly tied to use of brown coals or lignites, and to nuclear power. Very different, but in both cases serious, risks are connected to these energy sources. Hence, application of cleaner energy production technologies and energy saving are essential if economic growth and curbed environmental risks are to be combined.

The first generations of air pollution abatement policies in a country often focus on reducing peak concentration episodes. Inherently there are larger uncertainties connected to the possible long-term effects of sub-acute levels of air pollutants. An environmental policy with a long-term perspective should, however, aim at reducing the long-term average concentration level as well. Concerning ozone, for instance, history shows that local, short-term measures to reduce smog episodes often are relatively ineffective in reducing ozone exposure, and may actually also have the opposite effect under certain conditions (WHO, 1990).

In the following sections we have tried to assess the possible benefits from implementing measures that reduce the overall energy consumption, and thereby the general pollution level, in Hungary. These measures are described in the National Energy Efficiency Improvement and Energy Conservation Programs (NEEIECP). The concept of the program was elaborated by the Ministry of Industry and Trade and accepted by the government in April, 1994. The program constitutes the major part of measures to meet Hungary’s obligations under the Framework Convention on Climate Change (Poós, 1994; Pálvölgyi and Faragó, 1994; OECD/IEA, 1995).

Very briefly the main goals of the energy savings program are to:

- improve environmental protection;
- reduce the dependency on imports;
- save domestic energy resources;
- postpone the construction and installation of new base load power plants;
- increase the competitiveness of the economy;

- adjust to the energy policy of EU and to the OECD/IEA recommendations.

Two targets for medium range (5 years and 10 years) have been assessed. Because of the big economical uncertainties only the 5 years target is considered in the following. This scenario had the following key assumptions:

- the annual growth rate of GDP was expected to decrease up to 1995. Beyond 1995 the annual growth rate was assumed to increase by 1-2 %; (In 1995 the growth rate was ca. 1,4%, 1996: 0,8-0,9%)
- the price system of energy carriers should reflect realistic expenditure and the cross financing should be stopped;
- energy awareness should be developed as a consequence of rise in prices of the energy carriers;
- centralized subsidy and international aid programs (e.g. PHARE) should be assisted through a soft loan system.

The following estimates were given for the scenario:

Saved energy: 63.7 PJ/year
Saved energy cost: 373 mill. US$/year

Whereas the saved energy (in terms of PJ) is allocated on the various sectors and measures, the estimated total investment (capital and operating costs) needed during the implementation period of 5 years is an aggregate, the present value being 422 mill. US$ (i.e. the program seems to be highly profitable). We had some information on the estimated relative needs within some sectors, but it proved difficult to obtain a comprehensive picture of the cost estimates. This weakness of the program, as well as large uncertainties in the estimated energy saving potential, have also been pointed at by OECD/IEA (1995).

The various measures included in the NEEIECP are given in Table 1, and the possible energy savings within each sector are shown in Figure 2. It is important to note that the main single measure which gives 54% of the total assumed energy saving of 63.7 PJ, is "*energy awareness*", which is the energy economising expected to result mainly from increased energy prices. Other important measures are "*optimisation of the public transport system*", contributing 8%, "*reduction of energy consumption in vehicles*" and "*efficiency improvement of consumers equipment*", each contributing to 7% of the saved energy. Using emission coefficients for each sector (elaborated by Tajthy and co-workers, see Seip et al., 1995; Tajthy, 1993; and Tajthy et al., 1990), the corresponding reductions of air pollutants and greenhouse gases (GHGs) are estimated (Table 2) and used in the further calculations (see APPENDIX 1 for more details).

Table 1. Measures in the NEEIECP. Percentage reduction in total Hungarian energy consumption (in PJ) relative to 1992. EI=Energy efficiency improvement.

	Measure	**% of total PJ**
1.	Energy awareness	4.14
2.	Updating energy technologies (industry and agriculture)	0.30
3.	EI of energy prod. equipment	0.02

4.	EI of consumers equipment	0.54
5.	EI of energy transportation	0.36
6.	Red. of energy transmission/distribution loss	0.17
7.	Co-generation (heat/el.)	0.34
8.	Improved energy management in buildings	0.34
9.	Improving thermal insulation in industry	0.18
10.	Optimizing the public transport cooperation	0.60
11.	Reduced energy consumption in vehicles	0.54
12.	Renewable energy sources	0.12
	Total	7.65

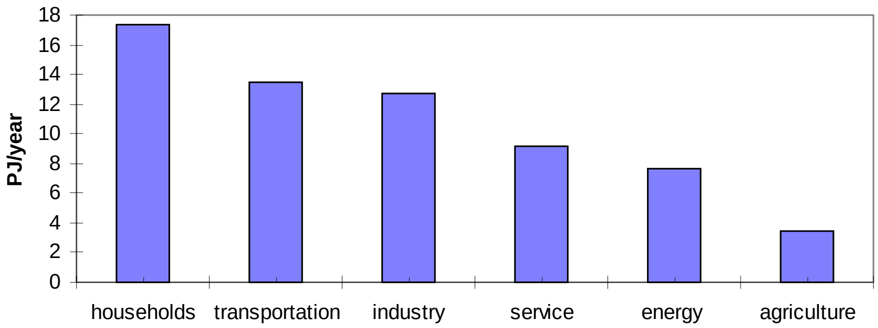

Figure 2. Possible energy savings if the short-term target (5 years) of NEEIECP were implemented (PJ/year).

Table 2. Reductions in annual energy consumption and emissions, estimated to result from implementation of the NEEIECP, relative to 1992.

	Reduction	**% of total**
Energy use (PJ)	63.7	7.7
TSP (ktons)	10.1	9.3
SO_2 (ktons)	46.8	5.7
N_2O (ktons)	0.5	7.8
CH_4 (ktons)	1.1	9.4
nmVOC (ktons)	5.8	10.0
CO (ktons)	71.8	12.3
NO_x (ktons)	17.4	10.1
CO_2 (ktons)	3800 - 4920[1]	5.8 - 7.5

[1] According to the Ministry of Industry and Trade (Poós, 1994) the reduction is 3800 ktons/year, which implies that it is assumed that most of the reduction relates to use of gas. The estimate of 4920 ktons/year is obtained by assuming that all energy carriers (gas, oil and coal) within the sectors influenced by the energy saving measures are affected.

In the following it is assumed that the energy savings primarily affect the consumption of fossil fuels. In reality this may not be strictly the case. The share of nuclear fuel of the total fuel consumption in public power plant (in PJ) is close to 40% (1992). Several measures within the NEEIECP are, on the other hand, related to sectors where nuclear energy is less important. When we estimated the reductions considering this aspect, the total energy saving became 10-15 % less (ca. 55 PJ/y). In view of other uncertainties, we found it justifiable to omit this factor in further calculations.

Another important assumption is that a given per cent overall reduction in the emissions of an air pollutant gives the same reduction in average concentration level in the cities. Although this assumption may be a reasonably good approximation on the aggregated level on which our calculations are done, it would not be valid for large reductions. In those cases contributions from the regional background concentration level (also caused by transboundary pollution) should be considered. Within a smaller range, the non-linearity should, however, not be too strong. For illustration purposes estimates are given for reductions up to 30% in Section 5.

Concerning the estimations of health effects we did, however, scale down the impact of reducing emissions in the agricultural sector due to the geographical distribution of these emissions. The same could also be done for damage to materials, but since the agricultural sector contributes very little to the SO_2-emissions, we did not make any adjustments here.

The underlying premise for the estimations is a status quo baseline scenario. That is, without implementation of the NEEIECP we assume that the emissions of air pollution do not change during the 5 year period. Looking at the current trend in energy consumption this does not seem to be an unreasonable assumption. A slight increase in the emissions in the baseline scenario would, in any case, not have altered the estimated benefits very much.

STATE OF HUMAN HEALTH IN HUNGARY AND POSSIBLE BENEFITS FROM REDUCING EMISSIONS

The life expectancy gap between East and West

Despite its limitations, mortality data provide the only basis for large-scale comparisons of health status, because it is the only health status measure collected routinely under similar conditions in different parts of the world. A conspicuous life expectancy gap has been evolving since the 1960s between CEE on the one hand and Northern and Western Europe on the other (for life expectancy in Hungary see Figure 3). The gap appears after a period of rapidly decreasing infant mortality in the CEE. In most of these countries the infant mortality is by now too low to influence life expectancy trends very much, and the gap is primarily due to differential survival starting at middle age (Hertzman, 1995). This is apparently also the case in Hungary, as can be seen from Figure 3b. The reduction in life expectancy for Hungarian men in the period 1970-1992 was largest, and nearly constant, in the age groups younger than 30, decreasing to zero around the age of 65, indicating that reduced survival in the age groups ca. 30 - ca. 65 is important for the trend. For women life expectancy has increased for most age groups. The jump from 0 - 1year of age is due to reduced infant mortality in the period (in 1970 life expectancy was higher at the age of 1 than at birth).

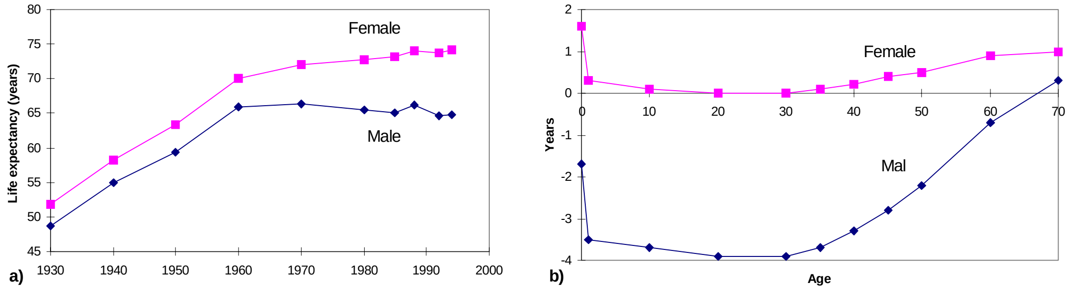

Figure 3. a) Life expectancy at birth in Hungary in the period 1930 - 1992; b) changes in life expectancy at different ages in the period 1970-1992 (Hungarian Central Statistical Office, 1993).

The physical environment is one out of many factors which may lead to the differences in health status observed across populations. For Central and Eastern Europe it has been estimated that up to 9% of the observed life expectancy gap may be explained by environmental pollution (Faechem, 1994). Life style factors, like smoking habits and diet, are probably the most important. Social and economic conditions, which for many people lead to deprivation and stress, may also contribute significantly. In Hungary, lifestyle factors obviously are decisive for reduced public health. For instance deaths due to cancer in the respiratory system increased from 2.8% of total deaths in 1970 to 4.7% in 1990; deaths due to cirrhosis of the liver increased from 1.1% to 3.7%; and deaths due to acute myocardial infarction increased from 8.7% to 9.9%. In this context it may seem of little relevance to focus upon ambient air pollution as a causal factor for reduced public health. However, it is well known that air pollution exposure will heighten the probability of premature death of individuals in advanced states of several common diseases. Hence, a rather small relative contribution of air pollution to the death rate for a prevalent condition may imply large absolute figures. For instance it is evident from epidemiological studies of mortality rates and air pollution that death rates due to respiratory and cardiovascular failure increase relatively more than the total rate (Derriennic et al., 1989; Wichmann et al., 1989; Schwartz and Dockery, 1992; Schwartz, 1994). As more than 50% of annual deaths in Hungary are due to cardiovascular failure, even a small change in the death rate would have large implications.

Air pollution concentration data

The air pollution exposure assessment was based on monitoring data from 1992/1993 for more that 90 cities and towns in all counties in Hungary. Data on 6 months' mean (summer and winter) levels of SO_2, NO_x, and dust fallout were available from all cities, whereas data for suspended particles were available mainly in the county capitals and Budapest. Average concentration levels for the cities were used, except for Budapest, where the figures are weighed according to concentration levels and population in the 22 districts (the exposure distributions for NO_2 and SO_2 are shown in Appendix 2). The majority of the larger and medium sized cities in Hungary and some smaller towns are included, representing totally ca. 5.4 million people, i.e. 52% of the population. According to the Central Bureau of Statistics in Hungary, approx.

37% of the population live in villages and rural areas. Thus, the cities from which we had data represent 83% of the urban population (57% for the TSP-data).

Figure 4 shows TSP- and NO_2-levels in some selected cities (discontinuous monitoring data). Concerning SO_2, the cities having levels above the Air Quality Guideline (see next Section) are with few exceptions situated in two counties, Komárom-Esztergom and Borsod-Abauj-Zemplén. Coal fired power plants and chemical industry are important pollution sources in these areas. The seasonal differences are generally much larger for SO_2 than for particles and NO_2, and the SO_2 guideline is not violated in the summer in any city.

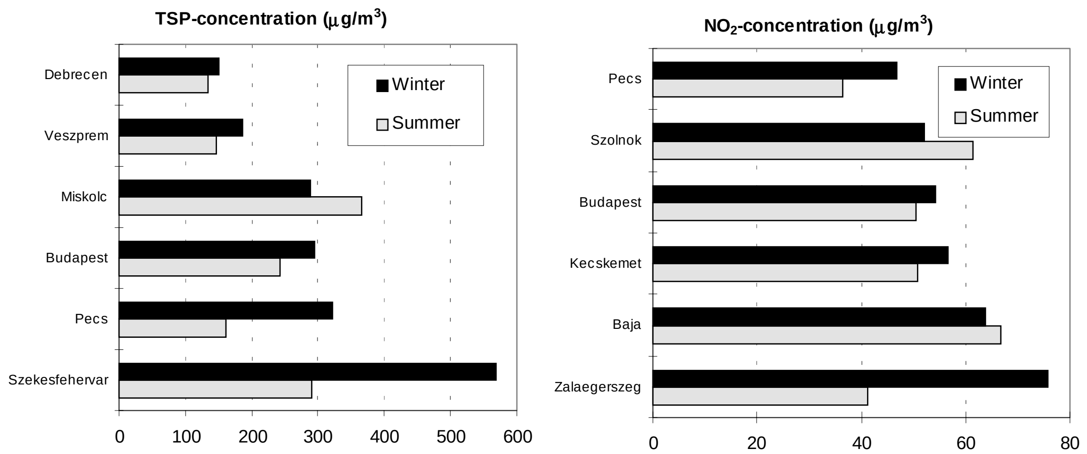

Figure 4. Seasonal mean TSP- and NO_2 -levels in some selected Hungarian cities.

Population exposure to air pollution

A rough indication of the seriousness of the pollution situation as regards health effects in a population may be obtained by estimating the number of people living in areas where air quality guidelines (AQG) are violated. In the following calculations we have used the Air Quality Guidelines for NO_2 and SO_2 established by WHO (1995), and the guideline for “settling dust”, i.e. dust fallout, given by Hungarian authorities (the most strict protection class). No guideline values are proposed by WHO for particulate matter, because there is no evident threshold for effects. However, to give an indication of the seriousness of the exposure level we have used the guideline for annual mean given by the Hungarian authorities (protection class 1, the second most strict class), which is 50 $\mu g/m^3$. Additionally, we have used the guideline for 6 months' mean of PM_{10} proposed by the Norwegian Pollution Control Authority, which is 40 $\mu g/m^3$ (SFT, 1992), and estimated a corresponding value for TSP using the estimated relationship between TSP and PM_{10} (see Section 0).

The AQGs used in the calculations are:

NO_2 :	40 $\mu g/m^3$
SO_2 :	50 $\mu g/m^3$
TSP:	50 $\mu g/m^3$ and 110 $\mu g/m^3$
Dust fallout:	12.5 g/m^2/30 days

Table 3 shows estimates of violations of AQGs for air pollutants monitored regularly in Hungarian cities. Estimates are made for the situation in 1992/-93 and if the energy saving program (NEEIECP) were implemented. The total population exposed to levels above AQGs was estimated taking into consideration the ratio between the number of people living in cities where monitoring data were available and the total urban population in Hungary. Since the towns for which we do not have data are smaller, we assumed that violations would not occur in all of them. Hence, for all components, except TSP, we added 10% and 20% to the number achieved from the cities where we had data, representing low and high estimates, respectively. For TSP we added 30% and 50% to obtain low and high estimates. The assumption that violations of long-term guidelines mainly occur in cities and towns, not in villages and rural areas, may imply that we are underestimating the numbers, especially for TSP. Generally, the uncertainties in the estimated figures for TSP are larger than for the other components, reflected by the broader range between the low and high estimates.

Figure 5 shows the estimated relations between reduced emissions (a proxy for reduced concentration levels) of different pollutants and the number of people living in urban areas where guidelines still would be violated (assuming the percentage reduction is the same in all areas). The data and calculation procedure used are the same as in Table 3 (the "high" estimates are used). Whereas reductions in the range of 30-40% would imply that the guidelines for NO_x, SO_2 and dust fallout probably would be attained in most urban areas, reduction of particles must exceed 60-70% to meet the AQG of 110 $\mu g/m^3$. To attain the 50 $\mu g/m^3$ guideline in these areas the concentration level probably must be reduced with more than 90%.

Table 3. Estimated number of people living in cities where long-term air quality guidelines (AQG) are violated (million people), 1992 and if the NEEIEC program were implemented.

Air pollution component	Pop. >AQG in cities with monitoring stations	Total for Hungary - 1992		Total for Hungary if NEEIECP is implemented	
		Low estimate	High estimate	Low estimate	High estimate
TSP > 50 $\mu g/m^3$	3.81	4.95	5.71	4.95	5.71
TSP > 110 $\mu g/m^3$	3.79	4.93	5.68	4.80	5.53
SO_2 > 50 $\mu g/m^3$	0.239	0.263	0.287	0.257	0.280
NO_2 > 40 $\mu g/m^3$	1.45	1.59	1.74	1.29	1.41
Dust fallout >12.5 g/m^2/30 days	0.58	0.64	0.70	0.47	0.51

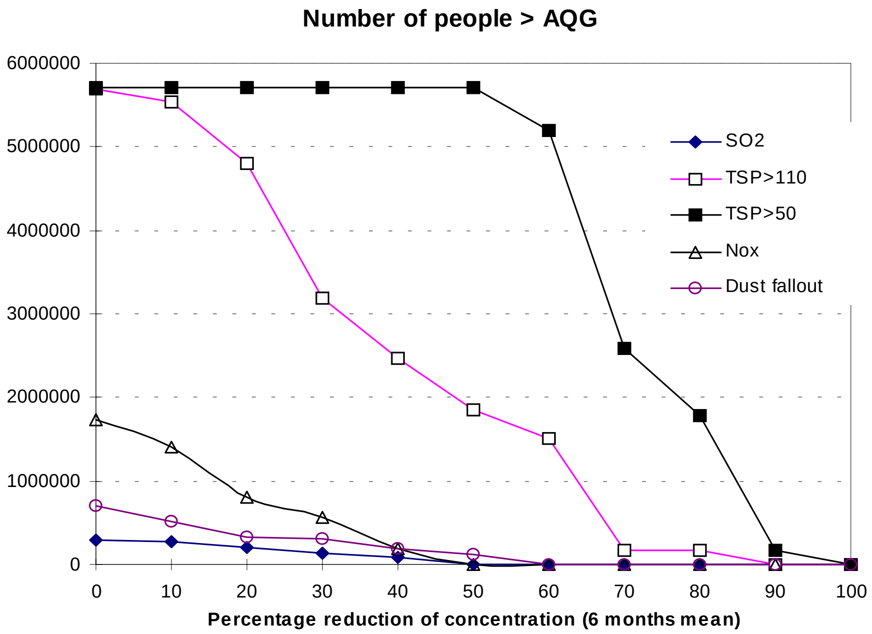

Figure 5. Relation between percentage reduction of concentration levels and number of people living in cities where air quality guidelines for SO_2, TSP, and dust fallout are exceeded.

Exposure-response functions from epidemiological studies

For an individual the exposure to air pollution may vary considerably over time. The indoor and outdoor micro-environment concentration levels vary according to e.g. the pollutant sources and dispersion patterns. A person's level of activity is among the factors determining the dose that enters the body. Additionally, the susceptibility varies among people, according to for instance age and health status. Hence, the risk of adverse health effects from air pollution is by no means equally distributed in a population.

Although some of the exposure-response functions for health effects and air pollution used in the following apply to specific groups, as elderly or children, in most cases they only provide estimates of average frequencies of health effects on a population basis. For instance there is no distinction between four persons having a one day illness episode and one person having a 4-day episode. It should be kept in mind that usually a subgroup of more susceptible individuals suffers most of the damage. However, the average per person estimates given below indicate the severity of the problem.

Epidemiological studies provide the best basis for establishing exposure-response functions for health damage in a population due to air pollution, because they generally apply to a cross-section of the population regarding age, gender, sensitive sub-populations, and also regarding the personal exposure level relative to the average pollution level. The exposure-response functions used here employ one indicator component for each effect type, and are mainly based on a review of epidemiological studies primarily from Western European countries and USA (Aunan, 1996). There are several problems connected to transferring risk estimates from one population to another, for instance we have to assume that the population-specific time-activity characteristics do not differ substantially. Whereas this is not likely to cause severe errors in our case, it is more problematic that the air pollution mixture (co-pollutants) may differ between Hungary and Western countries due to widespread use of coal in the former. Hence, use of one indicator component, found to be suitable in western studies, may lead to biased estimates in Hungary. For instance, as we shall see in the next Section, the PM_{10} fraction of suspended particles constitutes a smaller share of TSP in Hungary than what is often found in western studies.

Because we only had average concentration data for the Hungarian cities, and not daily (except for Budapest), the adjustment procedure suggested by Aunan (1996) was used to correct for the influence of daily variations on the effect estimates.

The different health effects associated with air pollution are usually determined by a combination of factors, and the challenge is to assess the importance of air pollution, i.e. what is the *excess number of cases* at different exposure levels and the present *attributable risk* due to air pollution. Using the methodology suggested in Aunan (1996), the excess frequency of an effect is the difference between the present frequency (empirical or estimated) and an estimated hypothetical baseline frequency, p_0, or, said in another way, the reduction in frequency which would result if one could reduce the pollution exposure with 100%. For effects having the daily concentration level as independent variable in the exposure-response function, the excess annual number of cases (deaths, symptom-days, or other end-points) is calculated as:

$$\textit{Excess annual cases} = \sum_{i=1}^{T} \left[\left(p_i(C_i) - p_0 \right) \cdot N_i \cdot 365 \right]$$

where:
T = the number of towns/cities included;
p_i = estimated present frequency in town/city *i*; a function of
C_i = the concentration level i town/city *i*;
p_0 = a hypothetical baseline frequency estimated for each effect (see Aunan, 1996);
N_i = population in city *i*

For effects having the annual concentration level as independent variable in the exposure-response function, *p* refers to the annual prevalence, and the factor 365 days is omitted. If data on the present frequency of an effect are available, a *city-specific* p_0 may be calculated, instead of using the more uncertain parameter p_0 derived from the epidemiological studies. Then, the attributable risk is indicated by the ratio between the estimated excess and the actual observed frequency, p^{obs}. We only had city-specific values for p^{obs} for mortality.

Correspondingly, the reduced number of cases resulting from a percentage reduction *x* in the concentration level, i.e. the health benefit, is calculated as:

$$\textit{Reduced annual cases} = \sum_{i=1}^{T} \left[\left(p_i(C_i) - p_{ired}(C_{ired}) \right) \cdot N_i \cdot 365 \right]$$

where:

$p_{i\,red}$ = the estimated new frequency in town/city *i*; a function of
$C_{i\,red} = C_i\,(1\text{-}x/100)$ = the estimated new concentration level

Relations between TSP, PM_{10} and NO_2.

Many exposure-response functions for health effects of air pollution relate to the concentration of suspended particles. Because particles are monitored only in a limited number of Hungarian cities (representing, however, 57% of the urban population), we investigated whether we could obtain reasonable estimates of the particle level from data on other pollutants which were available for more cities (representing 83% of the urban population). In Budapest both PM_{10} and TSP is monitored, whereas in the 18 county capitals only TSP is monitored. Using continuous monitoring data in inner Budapest during 1992-93 (183 observations of monthly mean), the following relation between NO_2 ($\mu g/m^3$) and PM_{10} ($\mu g/m^3$) was estimated (SE: Standard Error):

$$PM_{10} = 34.73\ (SE\ 3.18) + 0.60\ (SE\ 0.05) \cdot NO_2 \quad (R^2 = 0.40) \qquad (1)$$

The function probably overestimates the PM_{10} level when NO_2 is below ca. 10-15 $\mu g/m^3$. This, however, is rarely the case in Hungarian cities.

In order to test the NO_2 - particle function in the cities where both pollutants, NO_2 and TSP, are monitored, we needed to estimate a ratio between PM_{10} and TSP. This may be obtained from the calculated 6 months' mean of the continuous PM_{10}-data, and data on 6 months' mean of TSP (discontinuous monitoring) in Budapest (4 observations). Assuming that the continuous monitoring data generally are lower than the discontinuous (daytime) by a factor of 0.7 - 0.8[3] the following relation was estimated:

$$PM_{10} = 0.36\ (\pm 0.02) \cdot TSP \quad (R^2=0.39) \qquad (2)$$

The ratio indicated is lower than what is often found in studies in USA, where 0.5 - 0.6 is suggested as a conversion factor if no other data are given (US-EPA, 1982). The fact that the measuring points for PM_{10} in Budapest are situated 1.5 m higher above ground than the TSP measuring points could indicate that the actual ratio is somewhat higher than given in Eq. 2, but this probably explains only a minor part of the discrepancy with the US studies. Other studies have also indicated that the PM_{10}/TSP -ratio is lower in CEE than in Western Europe and the US (Clench-Aas and Krzyzanowski, 1996).

The R^2 value in equation 2 is rather low. If a constant was introduced in the function for PM_{10} and TSP, R^2 increased significantly:

$$PM_{10}=29.7(SE=7.2)+0.21(SE=0.04) \cdot TSP \quad (R^2=0.94) \qquad (3)$$

This indicates that a *linear* function is valid only within a certain range. Generally, when the measured TSP-level in a city was below ca. 120 $\mu g/m^3$, the TSP-concentration that we estimated using Eq. 1 and 3 was closer to the real, whereas

[3] This was based on data from Norway (Norwegian Institute of Air Research, L.O.Hagen, pers. comm.)

equation 2 gave better agreement when the actual TSP-level was higher. The agreement was, naturally, best in those cities where the NO_2/TSP-ratio is close to what it is in Budapest, which is approximately 0.2. In cities with very low NO_2-levels, combining Eqs. 1 and 3 would give a negative TSP-estimate.

Concerning the health response estimates that may be obtained by using the various equations, there is a relatively good agreement. Figure 6 shows the results for estimated reduced annual deaths related to implementation of NEEIECP. The estimates obtained by using PM_{10} from measured NO_2 is maximum 14% lower than those obtained by using PM_{10} from measured TSP by Eq. 3, and maximum 7% lower than those obtained by using PM_{10} from measured TSP by Eq. 2. Hence, our method probably at the most underestimates the response by 14%.

The annual mean NO_2-levels differ less between the cities than the TSP-levels. The predicted annual averages of particles in the 19 cities therefore appear to be less varying than the actual levels. To use the NO_2-data to estimate particle concentration may seem questionable in cities with high TSP-levels. For instance, in the industrial cities Miscolc and Székesféhevár, where the TSP-levels are very high, the NO_2 method gave an estimate of reduced annual deaths that was not much more than half of what was obtained by using the TSP data (using either Eqs. 2 or 3). However, this is less serious when we have in mind the purpose of the approximation procedure, which is to be able to assess possible benefits from energy saving measures. In cities with very high TSP levels process emissions from industry are an important source, and probably these emissions are less influenced by pure energy saving measures. The particle concentration estimated from the NO_2-data may simply be regarded as the level caused by combustion of fossil fuels.

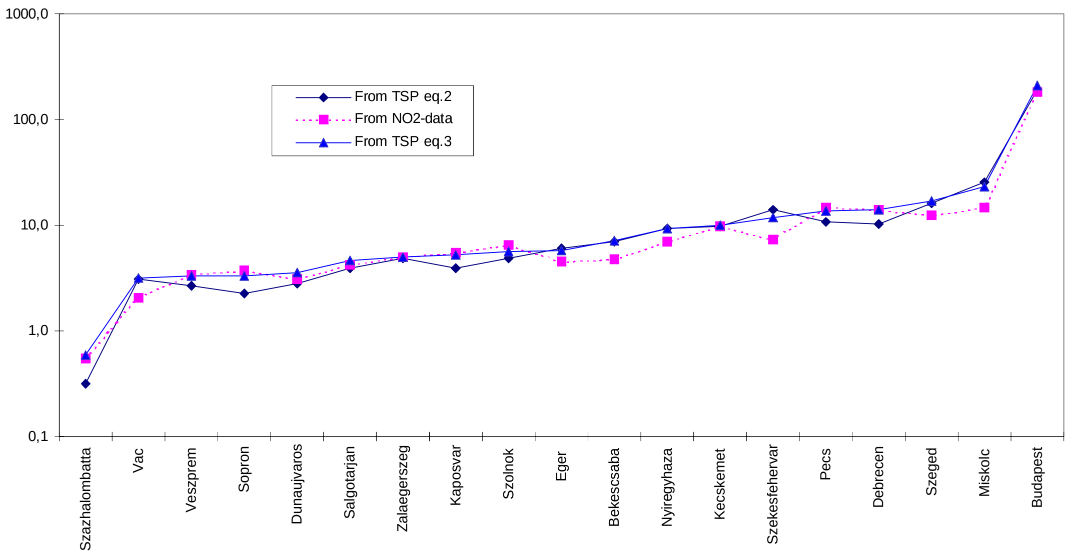

Figure 6. Comparison of estimates of reduced annual deaths in 19 cities and towns related to implementation of the NEEIECP, using PM_{10} estimated from measured TSP (equation 2 or 3) and PM_{10} estimated from measured NO_2.

The exposure-response functions we have used are summarized in Appendix 3. The uncertainty intervals given in Table 4 - Table 10 take into account the uncertainty in the conversion of NO_2-data into PM_{10}, the 95 CI (Confidence Interval) in the regression coefficient, the uncertainty in the hypothetical baseline frequency of the effect (p_0) (if this value is used), and the uncertainty in the conversion factor between various particle measure (if conversion is needed). In those cases where the exposure-response function originally related to TSP and is derived from studies in Western Europe and the U.S., we have used the "western" conversion factor of 0.5-0.6 to estimate a regression coefficient for a corresponding PM_{10}-relation. This is based on an assumption that PM_{10} is likely to be closer to the causal components of TSP. The resulting estimates are lower than they would have been if we applied the function directly upon estimated TSP-levels in Hungary (where a PM_{10}/TSP-ratio around 0.3-0.4 seems to be more correct). In light of the inconclusive air pollution epidemiology in CEE, we find the conservative estimates more reliable.

Mortality

The basis for establishing exposure-response functions for air pollution and mortality is more firm than for other health effects associated with air pollution. Several studies, mainly in the U.S., suggest that particles are the best single indicator for the air pollution species that affect mortality rates, and quantitatively similar relationships between particles and mortality have been reported over a large range of concentrations, in a variety of communities, with varying mixtures of pollutants and different climatology (see Aunan, 1996). There are also studies indicating that it may be justifiable to extrapolate results from western studies to other countries if epidemiological studies from the area of interest are not available (Ostro et al., 1995)

The reduced annual number of deaths estimated for a reduction corresponding to implementation of the NEEIECP is given in Table 4. It also gives the estimated current excess number of deaths. When we compare this to the approximate number of annual deaths in urban Hungary (accidents and violent deaths excluded), we arrive at a present attributable risk of 6.1% (95 % Confidence Interval (CI) 4.7 - 7.7%). Figure 7 shows the result of estimations for various reduction scenarios.

The actual death rates in the cities (accidents and violent deaths excluded) were used in the calculations, and the relative risk reductions were estimated from the exposure-response functions. We did not have age distributed mortality data for each city, but used the present average ratio between annual number of deaths in the age groups $\leq$ and $>$ 65 y in Hungary to make estimates for the two age groups.

Table 4. Estimated reduced annual deaths if the energy saving program were implemented, and present excess deaths due to air pollution in urban Hungary.

	Central estimate	95% CI
NEEIECP		
>65y	480	340 -610
≤65 y	70	50 -90
Present excess		
>65 y	5130	3630 - 6510
≤65 y	760	550 - 990

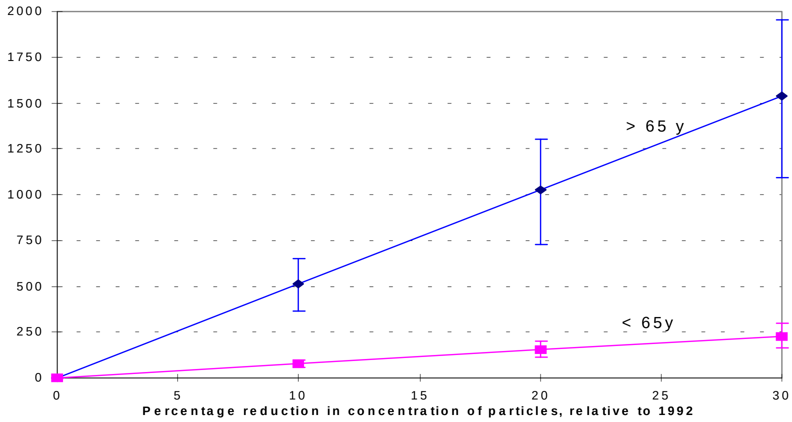

Figure 7. Estimated reductions in number of excess deaths (by age group - urban population) related to percentage reduction in emissions of particles relative to 1992. Bars indicate uncertainties as described in the text.

There are few studies on exposure-response relations for infant mortality. We have used the result from a study in the Czech Republic (Bobak and Leon, 1992). The function relates to PM_{10}. The function applies to deaths during the first year. Most deaths occur during the first week after birth, except the deaths due to respiratory diseases, which to a large extent occur after the age of 3 months (Hungarian Central Statistical Office, 1993). We did not have data on birth rate and infant mortality rate in each city. We used the average birth rate to estimate the number of births in each city, and the estimated baseline infant death rate and function from the Czech study to obtain approximate present death rates and possible future reductions. This externally derived baseline infant death rate is particularly uncertain.

The estimated annual reduction in infant deaths in urban Hungary predicted for various emission reduction scenarios, including implementation of NEEIECP, and present excess numbers are given in Table 5. According to the World Bank (1993) the infant mortality rate in Hungary in 1991 was 16 per 1000 live births. Application of this rate also to urban Hungary, implies approximately 1225 infant deaths annually in urban Hungary. Hence, implementation of NEEIECP would reduce this number with 2.8% (95% CI, 0.5% - 10.3%). The estimated excess deaths imply a present attributable risk of ca. 24% (95% CI, 5% - 77%). Intuitively, this seems somewhat high.

Table 5. Estimated reduced annual excess infant deaths (0-1 y) for various reduction scenarios, and present excess infant deaths due to air pollution in urban Hungary.

Reduction scenario	**Central estimate**	**95% CI**
NEEIECP	34	6 -126
10%	37	7 - 135
20%	72	14 - 259
30%	105	20- 373
Present excess	298	63 - 942

Other health end-points

Many *respiratory symptoms* are rather common, often have a viral etiology and may be associated with climatic conditions. On this background air pollution operates as a factor enhancing the susceptibility for infections and irritations, and prolonging and aggravating the symptoms.

The exposure-response functions for *acute respiratory symptoms* used here have daily concentration levels of particles as the independent variable and are based on European and US studies (see Aunan, 1996). The results for children (<14 y) and adults are shown in Table 6. Since we did not have any comprehensive statistics for the actual prevalence of acute respiratory symptoms in Hungarian children and adults, the estimated baseline prevalence was employed in the calculations. It was thus not possible to assess what percentage reduction the figures imply.

Table 6. Estimated reduced annual symptom-days (acute respiratory symptoms - average per individual) in children and adults in urban Hungary for various reduction scenarios, and present excess symptom-days due to air pollution. There are 1.235.000 children and 5.265.000 adults in urban Hungary.

	Central estimate	**95% CI**
Children:		
NEEIECP	1.0	0.2 - 1.9
10%	1.0	0.2 - 2.1
20%	2.0	0.4 - 4.1
30%	3.0	0.6 - 6.0
Present excess	8.9	2.0 - 16.7
Adults:		
NEEIECP	0.15	0.07 - 0.23
10%	0.16	0.07 - 0.25
20%	0.33	0.14 - 0.50
30%	0.48	0.21 - 0.74
Present excess	1.52	0.69 - 2.27

Pseudo-croup is a far more severe condition than the above symptoms, but also rather rare. Although fewer studies are available, the frequency of the disease has likewise been shown to be associated with air pollution (e.g. SFT, 1992). We used the study by Schwartz et al. (1991) in Germany to estimate the possible decrease in frequency of pseudo-croup from reducing emissions in Hungary. Since we did not have data on the actual frequency in Hungary, we have used the same methodology as above, using the estimated baseline incidence rate and the function. The annual average risk reductions for the scenarios are shown in Table 7.

Table 7. Estimated reduced annual symptom-days (pseudo-croup - average per child) in urban Hungary for various reduction scenarios.

Reduction scenario	**Central estimate**	**95% CI**
NEEIECP	$2.7 \cdot 10^{-5}$	$0.8 - 6.2 \cdot 10^{-5}$
10%	$2.9 \cdot 10^{-5}$	$0.9 - 6.7 \cdot 10^{-5}$
20%	$6.2 \cdot 10^{-5}$	$1.9 - 14.2 \cdot 10^{-5}$
30%	$9.9 \cdot 10^{-5}$	$3.0 - 22.5 \cdot 10^{-5}$
Present excess	$118.9 \cdot 10^{-5}$	$36.8 - 261.8 \cdot 10^{-5}$

Concerning *asthma symptom-days* among adults, we adopted the function proposed by Ostro et al. (1991) for the relation between daily level of $PM_{2.5}$ and response among adult asthmatics. Generally, the prevalence of asthma is found to be lower in CEE than in western Europe, see e.g. von Mutius et al., (1994). We did not have data for Hungary, and assumed that 3-6% of the adults are asthmatics. We also assumed a conversion factor between PM_{10} and $PM_{2,5}$ of 0.6 (uncertainty interval 0.5-0.7). The results are shown in Table 8.

Table 8. Estimated reduced annual asthma symptom-days (moderate or worse asthma days - average per asthmatic adult) in urban Hungary for various reduction scenarios.

Reduction scenario	**Central estimate**	**95% CI**
NEEIECP	2.40	0.04 - 4.75
10%	2.59	0.04 - 5.14
20%	5.49	0.09 - 10.88
30%	8.77	0.15 - 17.39

Chronic respiratory symptoms reported in epidemiological studies include illness frequency and symptom rates for i.a. bronchitis, asthma, and chronic wheeze and cough. The function for children used here builds on a relation found for bronchitis (Dockery et al. (1989), whereas the function for adults is based on three studies (see Aunan (1996) for a discussion). In the studies the criteria for the symptoms being chronic, were that bronchitis was diagnosed by a doctor, or that chronic cough was present at least 3 months of the year. Also here we used the estimated baseline prevalence and the functions, and as seen from Table 9, the function for children is very uncertain.

Table 9. Estimated reduced annual number of children and adults with chronic respiratory symptoms for various reduction scenarios, and present excess cases due to air pollution in urban Hungary.

Reduction scenario	Central estimate	95% CI
Children:		
NEEIECP	14040	360 - 76550
10%	15050	390 - 82250
20%	28520	770 - 159330
30%	40530	1150 - 228730
Present excess	96100	3860 - 499100
Adults:		
NEEIECP	16520	11300 - 27820
10%	17760	12150 - 29890
20%	34780	23830 - 58290
30%	51100	35060 - 85260
Present excess	147390	102410 - 238490

The exposure-response approach to damage estimation applied here is particularly difficult when it comes to assessing *cancer incidence* rates and the uncertainties are very large. We have taken another approach than above to make a rough assessment of the possible reduction in cancer cases which could result from reducing emissions. We assumed that around 5% of the present number of cases are attributable to air pollution. This is approximately what was found in a study in Krakow (Jedrychowsky et al., 1990) where the attributable risk due to air pollution was 10.5% in women and 4.3% in men. The TSP-level in the high pollution areas was above 150 $\mu g/m^3$. In a study in Texas, US, where the particulate pollution was considerably lower than in Krakow (below 85 $\mu g/m^3$ in the high pollution areas), it was estimated that 3% of the lung cancer mortality was attributable to air pollution (Buffler et al., 1988).

Totally, there are annually ca. 9200 new cases of lung cancer in Hungary (1992). In Western Europe and the U.S. comparisons of urban and rural dwellers indicate that the risk of getting lung cancer is higher by a factor around 1.5 - 2 in urban areas. If we assume that this is also the case in Hungary, it would imply that 6600-7100 lung cancer cases occur in urban areas (we did not have statistics for this). This implies an average rate in urban areas of $10.2 - 10.9 \cdot 10^{-4}$, which is 3.4-3.6 times higher than what is assumed as a background annual incidence rate in Western Europe and USA (300 lung cancer cases/mill.) (Hemminki and Pershagen, 1994).

A 5% attributable risk in Hungary then would imply that 330-355 cases (100 of which in Budapest) were attributable to air pollution in 1992. We estimated a relation between the long-term TSP-concentration and lung cancer incidence rate which would give a 5% reduction in the current total number of cases in urban Hungary if the ambient concentration of particles in each city hypothetically were reduced to zero. The function was obtained by estimating the number of cancer cases in each city where TSP-data were available, by using the average rate in urban areas weighted by the local TSP-level. The logistic regression coefficient (for TSP), assuming a current attributable risk of 5% (2%-8%), is 0.0003 (0.0001 - 0.0005), hence a linear increase per 10 $\mu g/m^3$ would be around 0.3% (0.1% - 0.5%). The estimated baseline incidence rate in urban Hungary, p_0, is $10.03 \cdot 10^{-4}$ ($9.71 \cdot 10^{-4} - 10.37 \cdot 10^{-4}$), i.e. still 3.3 times the Western background. The reduction scenarios are given in Table 10. Because the function was based on a study in an environment where the PM_{10}/TSP-ratio probably is close to what it is in Hungarian cities, we did not convert it into a PM_{10}-function (Eq. 2 was used to estimate the TSP-level in the cities). However, it may be that the amount of carcinogen components (e.g. PAHs) is higher in the Polish study area than in the average Hungarian city, hence the use of TSP as indicator component may be questionable.

The function for TSP and lung cancer rates rendered in Appendix 3 is based on the study in Krakow and a study in six U.S. cities (Dockery et al. 1993) (see Aunan, 1996). The two studies indicated very similar functions, but had both very broad uncertainty intervals. Using this function, we arrived at estimates indicating that implementation of NEEIECP in a long-term perspective would reduce the present number of annual cases with 4-6% and that the excess number represents 34-46% of the present number. Thus, it seems likely that this function overstates the response, at least when it comes to large reductions in concentration level. This also underscores the large uncertainties associated with transferring exposure-response functions from one area to another.

Table 10. Estimated reduced annual lung cancer cases, for various reduction scenarios.

Reduction scenario	**Central estimate**	**Uncertainty interval**[1]
NEEIECP	25	9 - 43
10%	27	9 - 46
20%	54	19 - 91
30%	81	28 - 136

[1]The uncertainty intervals reflect the assumptions about the present attributable risk being in the range 2%-8% and the urban/rural cancer risk ratio being 1.5-2.

The economic benefit of reduced health damage.

Economic value estimates for the health benefits were employed in order to make a tentative estimate of the monetised benefit from implementing the energy saving program. The unit value estimates are derived from Western studies (see US-EPA, 1995; Canadian Council of Ministers of the Environment, 1995; Krupnick et al., 1996). In these studies the willingness to pay (WTP) for health risk prevention is investigated by various methods (direct or indirect), or it is estimated from the cost of illness (COI). WTP is usually higher than COI because it includes a wider range of factors, and the WTP/COI-ratio, which is used to derive some of the unit values, has been estimated to be around 2 for many end-points. To estimate corresponding WTP values for Hungary we used the “relative income approach”, which means using the wage ratio between the US and Hungary to adjust the WTP values. In our case the relative wage approach implies a valuation multiplier of 0.16 (the average daily wage (1994) is \$15 in Hungary and \$93 in the US). The unit values and estimated benefits are given in Table 11.

The fact that WTP for health risk prevention is likely to take an increasing share of the budget as income increases, indicates that using relative incomes may overstate the WTP unit prices in Hungary. On the other hand, the wage level is decisive only for a part of the COI, and it may be that other costs may be relatively higher in Hungary than in the US, indicating that the relative wage income approach may understate the unit price in Hungary, if it is originally based on COI in the US. (For instance, the costs of hospital admissions and medication (embedded in the COI-part of some of the WTP-estimates) in Hungary are probably only to a limited extent a function of the wage level). A purchasing power parity index, which would have been another approach to transfer the benefit estimates, is not available for Hungary. Following the same argumentation, it would, if it could have been used, most likely have resulted in significantly higher benefit estimates.

Since the end-points in our study are not exactly the same as those valued by the US-EPA (1995), we have adjusted some of the estimates, and made some additional assumptions. The WTP for avoiding one case of infant death is assumed to be the same as for premature mortality in people ≤65 y. For cancer cases we used the calculation procedure proposed by the Canadian Council of Ministers of the Environment (1995), converting the estimate into US$. The survival rate has a large impact on the estimate. We have assumed a lower rate in Hungary than used in the Canadian study (we assumed a mean 5-year survival rate of 20% instead of 40%; this may still be too high, see e.g. Scientific American, 1996). The obtained unit price then became 30% higher.

To obtain unit values for impacts of respiratory symptoms, we assumed that 10% (an uncertainty interval of 5%-15% is used in low/high estimates) of the estimated acute respiratory symptom days (ARS) in Hungary are relatively severe and involve full activity restriction, i.e. a work day loss (see Aunan, 1996, for a discussion of this assumption). For the end-point denoted “restricted activity days” (RAD) by US-EPA (1995) it is assumed that 20% entail full activity restriction, hence we can not use the unit price directly. Instead we estimated a modified unit value for what we may call “ARS-restricted”, taking the daily wage multiplied by the WTP/COI ratio of 2 (our estimate becomes around 3 times higher than the RAD-value in US-EPA (1995). In addition to this, we have assumed that 0.5% (0.25%-0.75%) of our estimated ARS days involve a hospital admission (RHA), and applied the unit price proposed by EPA. For the remaining ARS days, we use the unit value given by EPA for “lower respiratory symptom days”, which are described as days where symptoms are noticeable but do not restrict normal activities.

Concerning asthma the estimated unit price proposed by US-EPA (1995) applies to a moderate asthma day, whereas the function used to predict the response in Hungary applies to a moderate or severe asthma day. Thus, using the value directly, as we have done, is conservative. Concerning the unit value for pseudo-croup in children we assumed that one case involves an emergency room visit (unit value from Krupnick et al., 1996) and two work days lost for one parent. This COI-estimate was multiplied with a WTP/COI ratio of 2 (see US-EPA, 1995). The severity of the chronic bronchitis cases in the basis study used by the EPA to estimate a unit value (a longitudinal study by Abbey et al., 1993 and 1995) is probably quite similar to the chronic bronchitis cases estimated for adults in Hungary, and the regression coefficients for the function for annual TSP-level and chronic bronchitis appear to be the same (see Abbey et al., 1993)[4]. Hence, we decided to use the unit value directly, additionally assuming that the value is applicable to chronic bronchitis in children as well. It is important to note that this WTP-estimate reflects the perceived welfare reduction of living with chronic bronchitis over the entire course of the illness, and is a measure of the present value of an effect which can span many years (as a minimum 3 months a year for at least two years). As seen in Table 11 benefits from reducing chronic bronchitis constitute about 75% of the total health benefit estimated here.

Table 11. Unit values (willingness to pay) for health impacts in the US and in Hungary (1994 US$), and estimated annual benefits from implementation of the NEEIECP in urban Hungary.

End-point	Unit value western studies[1]			Unit value adjusted for Hungary[1]			Benefit Hungary mill US$		
	Central	Low	High	Central	Low	High	Central	Low	High
Deaths>60y	3.4	1.9	6.8	0.548	0.306	1.097	261.2	103.5	663.4
Deaths<60y	4.5	2.5	9.0	0.726	0.403	1.452	51.1	20.7	133.6
Infant deaths	4.5	2.5	9.0	0.726	0.403	1.452	24.8	2.6	182.9
Lung cancer cases	3.0	1.7	6.1	0.490	0.273	0.979	12.8	2.5	43.4
ARS-Child-mild	11[2]	6[2]	17[2]	2[2]	1[2]	3[2]	1.9	0.2	5.5
ARS-Child-restricted	186[2]	93[2]	279[2]	30[2]	15[2]	45[2]	3.6	0.2	16.2
ARS-Child-HA	0.014	0.007	0.021	0.002	0.001	0.003	13.4	0.7	61.0
Pseudo-croup-tot	574[2]	473[2]	675[2]	93[2]	76[2]	109[2]	0.00	0.00	0.01
ARS-Adult-mild	11[2]	6[2]	17[2]	2[2]	1[2]	3[2]	1.3	0.3	2.8
ARS-Adult-restricted	186 [2]	93[2]	279[2]	30[2]	15[2]	45[2]	2.4	0.3	8.3
ARS-Adult-HA	0.014	0.007	0.021	0.002	0.001	0.003	9.1	1.0	31.1
Asthma days adults	36 [2]	13[2]	58[2]	6[2]	2[2]	9[2]	2.6	0.01	11.3
CRS-Child	0.24	0.14	0.38	0.039	0.023	0.061	541.2	8.1	4691.7
CRS-Adult	0.24	0.14	0.38	0.039	0.023	0.061	637.2	255.1	1704.9
TOTAL							1563	395	7556

[1] Mill. US$ unless noted
[2] US$

The relative wage approach was also applied by Krupnick et al. (1996) in their study of health benefits from air quality improvements in Central and Eastern Europe (CEE). They made an additional estimate for the mortality benefit, adjusting for the fact, discussed above, that the WTP for environmental quality does not necessarily vary proportionately with income. Concerning premature mortality some data are available that indicate an income elasticity (I_e) of 0.35 in the US for this end-point (i.e. reducing the risk of premature deaths is a good for which demand is more intense relative to income at lower income levels than at higher). Hence, Krupnick et al. (1996) adjusted the WTP value for mortality with the income elasticity found in the US in order to obtain a value applicable in the CEE-region. If we employ the same approach on our data, the total health benefit given in Table 11 would increase to approximately 2326 mill US$ (low and high estimates of, respectively, 682 and 9784 mill US$), and the benefit from reducing mortality would constitute 47% of the total health benefit.

[4] This is, by the way, an interesting example of a longitudinal study (of new incidences of chronic bronchitis) confirming the results (on prevalence rates) of several, generally assumed less reliable, cross-sectional studies.

However, in our view, it is probably unlikely that the demand function for reduced risk of premature death is log-linear over the entire span between the US wage level down to the levels in Central and Eastern European countries, as this adjustment procedure presupposes. More likely the function has an S-shaped lapse, implying that at low income levels marginal reduced mortality risk is a luxury good (i.e. $I_e > 1$), and, as wages increase, there is at some point a saturation taking place. The income elasticity of 0.35 found in the US may simply indicate that a saturation in the demand function has taken place. Thus, to approximate this (unknown) S-curve, it may be just as appropriate to assume a linear relation, as the wage-ratio approach does, as to employ an income elasticity found in the US.

DAMAGE TO MATERIALS IN BUDAPEST

Atmospheric corrosion and deterioration of materials is a cumulative, irreversible process taking place also in the absence of pollutants. The reactivity to various air pollutants varies greatly for different materials and pollutants. Together with the level of air pollution, particularly SO_2 and O_3, and the pH in precipitation, the deterioration processes also largely depend on meteorological conditions, especially the "time of wetness" (time fraction with relative humidity > 80% and temperature >0^0C.

More knowledge about deterioration processes and better methods for assessing stock at risk have increased the possibilities for better damage assessment of building materials in recent years. Figure 8 shows the steps in a model for calculation of cost of material deterioration (from Kucera and Fitz,1995). The excess costs for the different SO_2-concentration classes may be calculated by employing Equation 4. Hence, assuming that the necessary parameters and variables can be estimated, the cost savings resulting from reducing the SO_2-level to a lower class may be calculated.

$$K_a = K \cdot S\left[\frac{1}{L_p} - \frac{1}{L_c}\right] \qquad (4)$$

where
K_a = annual additional cost for maintenance/replacement per m^2
K = annual baseline cost of maintenance/repair
S = surface area of material (m^2)
L_p = maintenance/replacement interval in the polluted area (a specific SO_2-class)
L_c = maintenance/replacement interval in the clean area

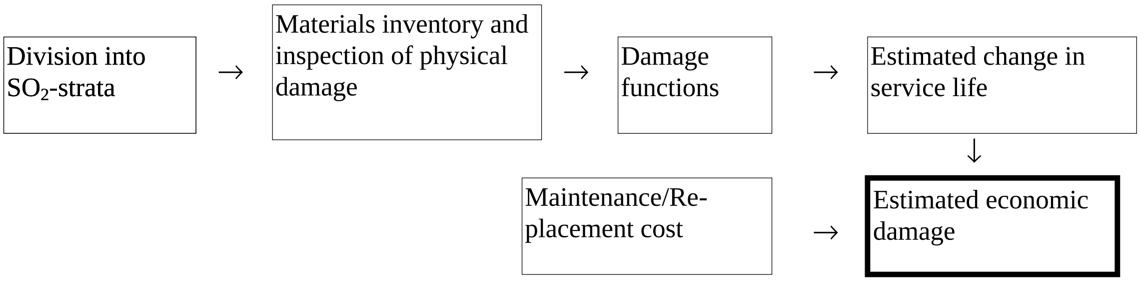

Figure 8. Main steps in a model for calculation of costs due to air pollution induced material damage (from Kucera and Fitz,1995).

The damage functions referred to in Figure 8 may relate the mass loss during a certain period, given in g/m^2, or the reduction in thickness, given in µm/yr, to the exposure level. Such functions are established for a number of materials, based on data from test-field in different countries (see Kucera and Fitz, 1995). In order to make an economic evaluation the damage functions have to be transformed into life-time, or service time, functions that can be linked to the maintenance and replacement costs. This implies that several assumptions have to be made as regards for instance average thickness and structure of the material in question.

A preliminary estimate of the economic loss due to damage of materials in Budapest was made based on statistics of building mass and materials in the city together with results from studies in other European cities (Kruse, 1995). Table 12 indicates the distribution of buildings in the city according to pollution level. The buildings were categorised, and for each building category the average size and surface of various material types were estimated. The estimates of material types were based on studies in other European cities. This implies relatively large uncertainties. Life-time functions from the international literature were employed. Whereas recently proposed functions for many materials has SO_2 and O_3 as independent variables (e.g. Henriksen and Hågenrud, 1995 and Kucera and Fitz, 1995), functions employing only SO_2 were used in Budapest, due to the lack of sufficient O_3-data. For instance, the life-time function employed for unpainted, galvanised steel was:

$$V_{corr} = 0.29 + 0.039 \cdot [SO_2]$$

where

V_{corr} = corrosion rate for zinc, µm/year
$[SO_2]$ = SO_2 -concentration, $\mu g/m^3$

Using 1990 as the baseline year, the estimated annual saving in total corrosion costs if average SO_2 levels were reduced to less than 20 $\mu g/m^3$ in all regions of Budapest, was about US$ 50/inhabitant, which implies a total annual saving of US$ 100 mill. Corresponding figures for Prague and Stockholm, are US$ 110/inhabitant and US$ 20/inhabitant, respectively (Kucera et al., 1993).

In 1990 an area representing 10-20% of the city area (in five districts in Budapest) had annual mean SO_2-concentration above 20 $\mu g/m^3$. The level had at that time been steadily decreasing for several years, and it has also decreased somewhat since then. An estimated SO_2-concentration below 20 $\mu g/m^3$ in all districts except one may be obtained by imposing a flat 25% reduction for the city as a whole. At 35% reduction, all districts should have an annual average SO_2 -level below 20 $\mu g/m^3$. A flat 6% reduction, which is the estimated average SO_2-reduction resulting from implementation of NEEIECP, would reduce the area having a SO_2-level above 20 $\mu g/m^3$ with 20-25%. If we take into account the building density in the various districts, we arrive at a reduced annual cost in the range 30-35 mill US$.

Table 12. Distribution of buildings according to pollution level in Budapest (from Kruse, 1995). SO_2 -levels for 1990.

	SO_2 class		
	< 20 $\mu g/m^3$	**20 - 60 $\mu g/m^3$**	**> 60 $\mu g/m^3$**
Number of buildings	102 540	689 570	66 840
Area (km^2)	112	401	12

DAMAGE TO CROPS AND NATURAL VEGETATION

It is assumed that ozone is the component of most concern as regards crop production, and it is dispute in the literature as to whether SO_2 does serious damage to crops at present levels in Europe and the US (CEC/US, 1993). Critical levels for ozone were first defined in 1988 (UN-ECE, 1988) in terms of threshold concentration levels for specific time periods. The concept has been further developed and in order to take into account the importance of long-term accumulated exposure, it is recommended that so-called AOT40 values (*a*ccumulated exposure *o*ver the *t*hreshold of *40* ppb) are used (UN-ECE, 1994; Kärenlampi and Skärby, 1996). AOT40 (unit: ppb-h) is calculated as the sum of the difference between the hourly ozone concentration in ppb and 40 ppb for each hour when the concentration exceeds 40 ppb. Critical levels for agricultural crops and forest trees (deciduous and conifers) are given in Table 13. The value for crops are based on data for wheat, but until further data are available for other crops, it is recommended that this value is applied to all crops. There are large uncertainties connected to the use of AOT40 values as critical levels, and the concentration of 40 ppb does not represent an absolute threshold level of effects. Several factors, other than the ozone level, are decisive for possible crop losses. For instance, the actual O_3-dose that passes the stomata is highly dependent on climatic conditions. To what extent a given dose is adverse also depends on the life cycle phase of the plant. The least deviation from a 100% yield that is statistically significant is a 4-5% loss, and the critical level proposed for crops is therefore based on a 5% crop loss. An isolated short-term incidence of visible injury may affect the economic yield of a crop whose value depends on the physical appearance, and short-term critical levels are therefore defined for visible ozone injury on crops (two levels are proposed, depending on the air humidity).

Table 13. Critical levels (AOT40) for agricultural crops, natural vegetation and forests (Kärenlampi and Skärby, 1996).

Receptor	**Critical level for damage (ppb-h)**
Agricultural crops and natural vegetation	
- 5% yield loss[1]	3000
- Visible ozone injury[2]	500[3]
	200[4]
Forest trees[5]	10000

[1] Resulting from long-term exposure; AOT40 is calculated for daylight hours, and for three months; soil moisture is not limiting; [2] AOT40 is calculated over 5 days, for daylight hours, and for three consecutive days; [3] For high vapour pressure deficit conditions; [4] For low vapour pressure deficit conditions;[5] Resulting from long-term exposure; AOT40 is calculated for daylight hours during a 6-month period.

The mean daily maximum concentration of ozone in the growing season (April - September) has been estimated to be 120 - 140 $\mu g/m^3$ in Hungary (for 1990) (EMEP/MSC-W, 1996a). The long-term critical level defined by UN-ECE in 1988 was 50 $\mu g/m^3$. Calculated AOT40 (for crops) in Hungary is 10000-20000 ppb-h, i.e. about 3-7 times the critical level for 5% crop loss (EMEP/MSC-W, 1996a; we use the estimate obtained by assuming a deposition velocity of 0.8 cm/s).

The values given above are so-called Level I critical levels. UN-ECE recommends that any estimated exceedance should only be used as an indication of the degree of risk, and should not be converted into an economic yield loss estimate (Kärenlampi and Skärby 1996). To obtain such estimates, it is stated, a level II approach, taking into consideration several climatological, biological and chemical factors, should be developed. So far, methods for adjusting yield loss estimates for soil moisture availability have been proposed, and in dry years, or in dry areas, the estimates of yield loss due to ozone become significantly reduced (Führer, J., 1996). Atmospheric vapour pressure deficit and atmospheric conductivity are other factors that may be important in controlling the ozone damage.

As a preliminary approach, in lack of a more comprehensive methodology for a level II approach, we make some tentative estimates of the possible crop losses in Hungary using the linear relationship between the relative yield of wheat and AOT40 that has been demonstrated. Several data sets indicate a linear exposure-response function up to about 35000 ppb-h . The linear function is approximately (Führer, 1996):

Yield loss (%) = 0.0017 $\cdot$ AOT40 (ppb-h)

In the case of Hungary this function implies a crop loss of in the range 17%-34%. To indicate the possible annual economic impact on cereal crop production, we have made estimates for a crop loss in this range, see Table 14. The production of cereals has been fluctuating the last years, and the average production in the period 1990-1992 was used in the calculations. The average 1992 production prices (the weighted average of procured and market price, in HUF), and the world reference prices (unit export value for non-rouble trade, in US$) were used. The estimates do not take into account demand elasticity (how the price per unit product responds to a change in the production). This probably would have an impact when the yield loss is of this magnitude.

The exposure-response function is highly uncertain beyond 20% loss, and one should probably have more confidence in the lower range of the estimated loss. Moreover, possibly a relatively dry summer climate and lack of irrigation (only about one third of the crop production area is irrigated) indicate that the function may overstate the crop loss in Hungary. If we use the function for grain yield loss under non-irrigated (dry) conditions rendered in Führer (1996) the obtained crop loss due to ozone in Hungary becomes 11-21%.

Table 14. Estimated present annual economic loss due to ozone induced crop loss.

Crop	Loss (1000 tons)	Loss (bill. HUF)	Loss (mill. US$)
Wheat	1090- 2730	7.8 - 19.4	114 -283
Rye	40 - 100	0.3 - 0.7	5 - 13
Barley	320 -810	2.1 - 5.2	41 - 102
Maize	1160 -2900	8.8 - 22.1	127 - 316
Rice	5 - 15	0.0 - 0.1	1 - 2
Total	2620 - 6550	19.0 - 47.5	287 - 716

The atmospheric chemistry behind the formation of ozone is non-linear. Hence, it may be difficult to estimate the reduction in the O_3-concentration resulting from reduced emissions of the two precursors VOCs and NO_x. This complicates the development of abatement strategies. Generally, it is more efficient to reduce NO_x -emissions in low-NO_x-areas, and more efficient to reduce VOC-emissions in high-NO_x-areas. Anyway, an implementation of the NEEIECP in Hungary alone will have no significant impact on the crop loss. For instance the estimated reduced AOT40 (for crops) resulting from 30% reduction in NO_x-emissions is 625 ppb-h, whereas the corresponding reduction resulting from 30% VOC reduction is only 54 ppb-h (EMEP/MSC-W, 1996b). Thus, an emission reduction considerably higher than in the NEEIECP, would reduce the present crop loss with, at the most, one percent point. However, the EMEP calculations shows that significant increases in crop yields are likely to be obtained if NO_x and VOC emissions are reduced in large regions in Europe.

Forests in large parts of Europe are probably adversely affected by air pollution although the understanding of the causes and mechanisms is poor except in the most polluted areas where direct effects are plausible. The damage in Hungary is considerably less than in Poland and the Czech and Slovak republics and has shown no significant trend since 1990 (UN-ECE/CEC, 1996). We have therefore not tried to quantify the forest damage due to air pollution in Hungary. Any effects of general reductions of the size estimated for NEEIECP will be small. However, in areas with the highest present SO_2 values there may be improvements in areas close to pollution sources.

CLIMATE CHANGE

The damage assessment in this paper has focused on the possible benefits on a local and regional scale of curbing air pollution. However, energy saving will also reduce the emissions of several greenhouse gases (GHGs). Implementation of NEEIECP should for instance reduce CO_2 emissions by 5.8 - 7.5%, see Table 2. According to IPCC (1996) the uncertainties in the present simulations of regional climate change are still too large to yield a high level of confidence in simulated scenarios. Most models indicate that Hungary is situated in an area which will experience a warmer climate. How the precipitation pattern may change and whether the climate is likely to be dryer is unknown. Large areas in Hungary, especially the puszta, have a relatively dry climate, hence even a moderate warming and/or reduced precipitation could have a significant negative effect, i.a. for the agricultural production. Of course, even more for greenhouse gases than for tropospheric ozone, isolated reductions in Hungary are of minor importance.

The importance of sulphate aerosols as a cooling agent has recently been a subject of much concern. Model simulations have indicated that reductions of atmospheric sulphur components will enhance the global warming. The measures discussed in this paper reduce all components and thus should be non-controversial. However, efforts to specifically abate SO_2 and particles would certainly be beneficial in Hungary. In our view the status of knowledge of the interactions between a possible global warming effect and regional effects of aerosols is not sufficient to justify an avoidance of sulphur reduction measures. This is not only because of the large positive local and regional effects resulting from sulphur reductions, but also because a regional cooling is not necessarily beneficial, e.g. it may lead to unfortunate changes in the atmospheric transport patterns (IPCC, 1996).

THE COST OF ENERGY SAVING

The unit cost of implementing energy saving measures within the various sectors may be given in terms of the cost per reduced emission unit or per reduced damage unit, and a ranking of the measures may be based on either of these. We were able to make detailed estimates of the emission reductions resulting from each particular measure within each of the six economic sectors, and concerning the health effects we were able to establish a corresponding benefit matrix. Detailed information on the investment and operating cost for each measure was, however, not available, hence cost/benefit-ratios could not be calculated.

However, we wanted to indicate what kind of estimates that can be made, if more detailed information is available; estimates that are helpful in prioritising investments across sectors and various measures. For illustration purposes we have used the investment estimate (present value for the 5 year implementation period) given in Section 4, in Figure 9 and Figure 10. If we assume that the energy saving (in PJ) obtained from a given investment is the same in all sectors (which is probably *not* the case), the estimated unit costs for emission reductions varies simply according to the different emission coefficients across sectors. Figure 9 shows the cost per kton of emission reduction within each sector for the three main air pollutants, given this assumption. For instance, TSP-reductions seem to be more expensive in the industry and energy sector, consistent with the fact that an extensive electro-filter program has already been implemented in public power plants. Concerning NOx-reductions it seems to be cost-effective to concentrate on the transport sector, whereas for SO_2-reductions the energy and industry sectors should be prioritised. Concerning CO_2 there are no large differences between the sectors, except that the unit cost seems to be somewhat smaller in the energy sector due to a higher emission factor. For the other main greenhouse gases there are large differences between the sectors. If reliable cost estimates had been available, a ranking of measures within each sector could have been made based on an optimisation, taking into consideration several components.

Figure 10 illustrates the benefit/cost-ratios of implementing energy saving measures within each sector, regarding only the health benefit (see Table 11). We assumed that the effect of the energy saving measures lasts for 15 years, and that the estimated annual health benefit is constant over this period. The present value of the health benefit is calculated using a discount rate of 6%. According to the resulting figures abatement measures within the household and service sectors should be prioritised, if reduced public health damage is a main policy goal. The net benefit follows the same pattern, except that the industry sector has a higher net benefit than the energy sector.

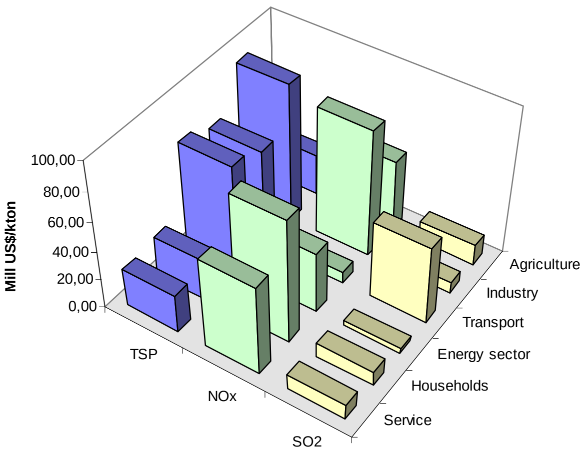

Figure 9. Mill. US$ per kton reduced emission of TSP, NOx and SO_2, assuming the same investment per PJ saved energy in all sectors.

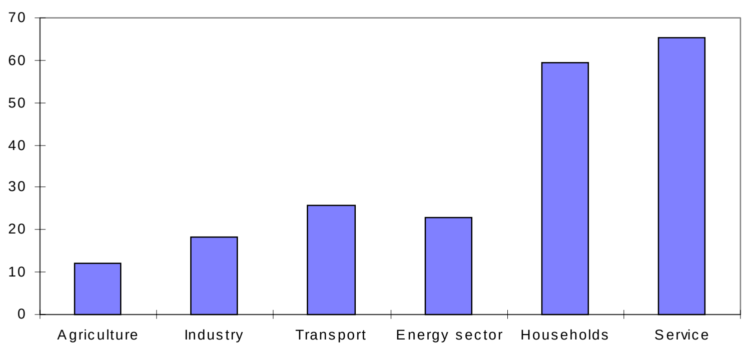

Figure 10. Tentative benefit/cost-ratios: Health benefit resulting from reduced emissions versus the investment in each sector.

TOTAL COSTS AND BENEFITS - UNCERTAINTIES AND SOME TENTATIVE CONCLUSIONS

Concerning the estimated health benefits from implementing NEEIECP there are three main sources of uncertainties. First, the limited monitoring data and the fact that we had to estimate the level of particles in several cities introduce uncertainties (see Section 5.5). Relative to other uncertainties, this may, however, be the smallest. The next step, which involved transferring exposure-response functions from western epidemiological studies, probably introduces larger uncertainties, i.a. because the health status differs between the populations and the pollution mixture is different. The uncertainties, both in the original function and those arising due to the transferring, vary between the different health end-points, partly reflected in the width of the uncertainty intervals. Probably, the mortality estimates are the most reliable. The most critical uncertainty relates to the estimates of reduced prevalence of chronic bronchitis, especially in children. Generally, the strong reliance on particles as indicator component in the exposure-response functions used here may seem questionable in the light of recent European studies. These tend to be less concordant in the sense that whereas some find association with particles other find closer associations with SO_2, ozone, and/or NO_2 (e.g. Bacharova et al., 1996; Touloumi et al., 1996; Spix and Wichmann, 1996).

Concerning the estimates of present excess cases, these are far more uncertain than the estimated benefits of small reductions, and only serve as very rough indications. This is because a wider range of the exposure-response function is used and because a possible threshold level is ignored (the functions originally apply to marginal changes). There are also indications that exposure-response functions derived in western studies may overstate the response in areas with high pollution levels, a fact that adds to the uncertainty of our estimates. Some analyses in areas where higher levels occur report that the relationship becomes flatter as the pollution increases, being consistent with a logarithmic function (Spix et al., 1993, WHO, 1995). The logistic functions applied here imply the opposite, i.e. the greatest benefit per unit of emission reduction is obtained for the initial measures (though the curvature is in most cases rather small in the relevant range).

The third large source of uncertainty relates to the benefit transfer, i.e the estimation of unit prices in Hungary using U.S. studies[5]. As this uncertainty relates not only to the estimates, but to the methodology upon which they are based as well, it is difficult to quantify (the uncertainty intervals given above only relate to the original unit price estimates and the adjustments we made). Concerning the valuation of chronic bronchitis, for which the largest benefit seems to occur, the estimates are highly sensitive to the assumptions about the average severity of the cases. For instance, assuming that the degree of severity is about half of what is assumed in this paper (3 on a scale from 0-13, instead of 6.5), would strongly reduce the estimated total health benefit, and the benefit from less bronchitis would constitute 44%. Thus, despite large uncertainties, it is evident from our analysis that changes in the prevalence of chronic diseases are very important, not at least from a socio-economic point of view. This has been pointed at by several, but few attempts have been made to quantify it (Portnay and Mullahy,1990). In a study in Oslo, however, it was estimated that 37% of the total socio-economic cost of the present level of particulate air pollution was related to chronic respiratory morbidity, whereas 54% was related to premature mortality (Rosendahl, 1996).

A fundamental problem with this kind of analysis is the limited possibilities for validation of the health effect estimates. However, a few epidemiological studies are available that may be used to test our estimates of the present frequency of effects made in those cases where comprehensive statistics was not available. For instance, in a study in Százhalombattá (Tar and Tajthy, 1991) the average frequency of respiratory illnesses in adults (30 - 50 y) during 1988-1989 was found to be 4.7% for men and 7.1% for women in those who were only living in the city, and 11.6 % and 12.3% in men and women, respectively, who both lived and worked in the city. In our calculation of acute respiratory symptoms Százhalombattá came out with a present average frequency in adults of 3.1% (2.7%-3.5%). Even though the air quality in the city has been considerably improved since the health survey was undertaken, the comparison indicates that in this particular case using the exposure-response and the

[5] A comprehensive study in Europe, called "Benefits transfer and economic valuation of environmental damage in the European Union - with special reference to health", is being implemented under the Environmental and Climate Program (EU DGXII), but no results are yet available. Anyway, CEE is not included in the study (Ståle Navrud, pers. comm.).

baseline prevalence understates the total prevalence. The error in the estimate of possible changes in response due to TSP-reductions is, however, not known. Another study, in Sopron (Rudnai et al., 1996), gives account of the average weekly respiratory morbidity rate in children during 1990. This varied between ca. 3% - ca. 12% in the various districts, and was on average for children 1-14 years ca. 6% in those parts of the city which had a moderate pollution level (SO_2). In our calculation Sopron was estimated to have an average frequency of ARS in children of 9.7% (7.4%-12.2%), i.e. quite close to the reported rate.

The benefit concerning damage to materials is likely to be considerably lower than for health. However, the estimate given here only comprises buildings in Budapest and does not include an evaluation of the esthetic and cultural dimension. Hence, a comprehensive assessment for Hungary would give a considerable higher estimate than the estimate from Budapest given here extrapolated to the whole country. The largest uncertainties in the estimate of reduced replacement and maintenance costs probably relate to the assumed distribution of various materials and the costs estimates (Kruse, 1995).

In this paper we have analyzed possible environmental consequences of implementing an energy saving program that has been elaborated by Hungarian authorities. Whether the elements in the program actually will entail the predicted emission reductions is, however, uncertain, as the sector analyses behind the predictions have a limited accuracy. Further studies of the costs of the specific measures proposed in the program would be beneficial, as the present lack of accurate data limits the possibilities for performing the detailed analysis needed in setting priorities in the air quality management and in environmental policy making in general. Also, studies on exposure-response relations and valuation of environmental amenities, performed in Hungary in the present situation, would improve the accuracy and quality of the analysis. The hazy prospects concerning economic activity and energy demand even in the near future, however, inherently complicate forecasting and imply large uncertainties.

Our analysis indicates that the main benefit caused by reduction of concentrations of pollutants by implementing NEEIECP (or similar measures) relates to improved public health. The monetised values are highly uncertain due to limitations in data and methods. However, even the low estimate of the annual benefit, which is obtained by using the low estimates of all parameters in all steps of the calculation, would give a positive net benefit of the program (assuming total investments as estimated by the Hungarian Ministry of Industry and Trade). Even though the investments needed may be considerably underestimated, as indicated in Section 4, it is very likely that annual benefits of improved health conditions alone are larger than these investments. In addition there are significant benefits due to reduced damage of materials (30 - 35 mill US$ annually in Budapest only). The damage to crops due to ozone is large, but a significant improvement depends upon concerted actions in several countries. Although the recommendation to carry out the measures in NEEIECP is clear from our study, practical problems related to funding the program in the present situation of scarce capital access have not been considered. Neither have we discussed the specific measures included in the program. For instance, we would have thought that renewable energy, and especially thermal energy for which Hungary has got large resources, is worth developing in a more long-term perspective.

In this study we have emphasized local and regional effects although the measures will reduce emissions that may affect climate as well. It seems likely that there will be international agreements on CO_2 emissions in the near future. Since the CO_2 emissions in Hungary decreased by 26% in the period 1985 - 92, Hungary will probably not have serious problems to comply with such agreements in the first years. However, in the future curbing of these emissions will be important. Our analysis shows that measures that are cost-effective by considering local and regional effects, may also reduce CO_2 emissions considerably, underlining the advantage of an integrated approach in framing a cost-effective environmental policy.

Acknowledgements

We would like to thank Kristian Kruse for letting us use the data and results from his study of material damage in Budapest. We would also like to thank Dr. Tihamér Tajthy and Dr. Per E. Schwarze for useful comments on an earlier draft.

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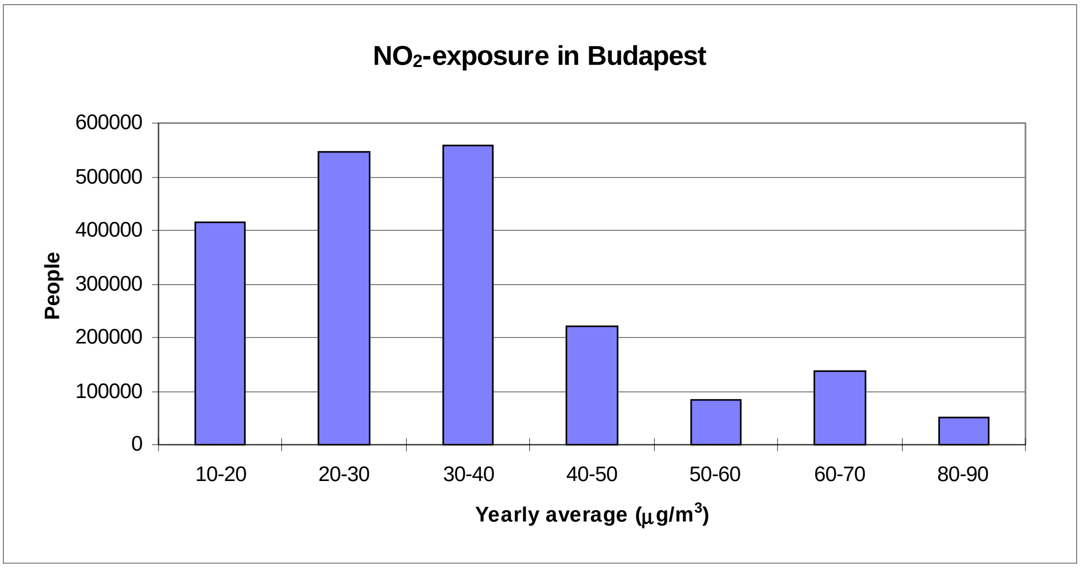
APPENDIX 1 *Estimated annual reductions in energy consumption and emissions (ktonnes) for the NEEIECProgrammes.*

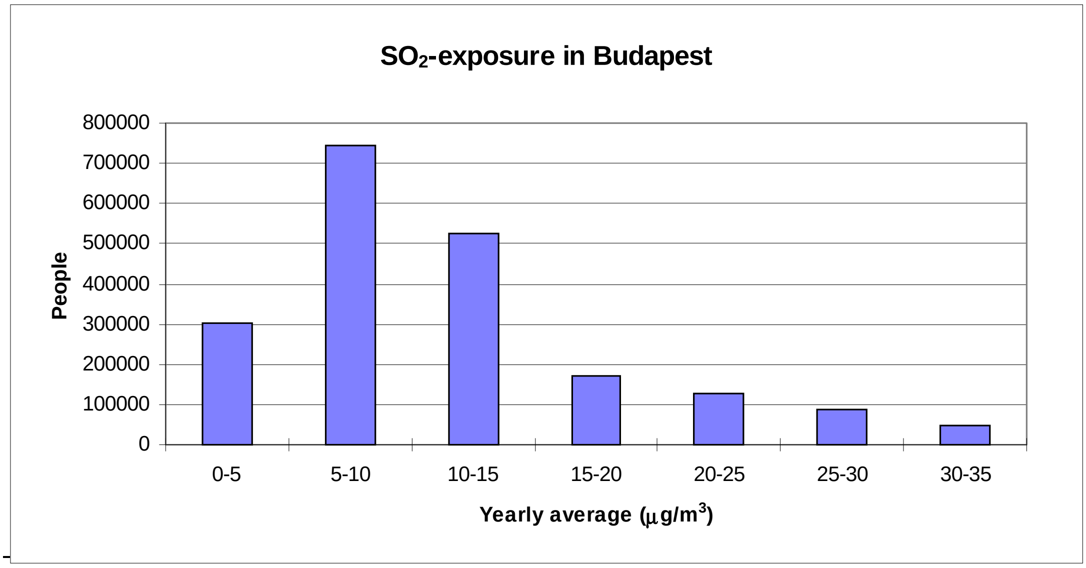
Sector	Measure	PJ	CO_2	CH_4	N_2O	Nox	CO	nmVOC	SO_2	TSP
households	1. Energy awareness	14	1098	0,35	0,09	1,10	5,52	1,87	9,32	3,26
households	4.EI consumers equipm.	2	157	0,05	0,01	0,16	0,79	0,27	1,33	0,47
households	8. Improved energy managem. in buildings	1	78	0,03	0,01	0,08	0,39	0,13	0,67	0,23
households	12. Renewables	0,3	24	0,01	0,00	0,02	0,12	0,04	0,20	0,07
service	1. Energy awareness	7	524	0,15	0,04	0,76	0,29	0,45	4,36	1,78
service	4.EI consumers equipm.	1	75	0,02	0,01	0,11	0,04	0,06	0,62	0,25
service	8. Improved energy managem. in buildings	1	75	0,02	0,01	0,11	0,04	0,06	0,62	0,25
service	12. Renewables	0,2	15	0,00	0,00	0,02	0,01	0,01	0,12	0,05
transportation	1. Energy awareness	4	291	0,05	0,03	3,69	18,78	0,45	0,50	0,40
transportation	10. Public transport	5	364	0,07	0,04	4,61	23,48	0,56	0,63	0,50
transportation	11. Red. energy consumption vehicles	4,5	327	0,06	0,03	4,15	21,13	0,50	0,57	0,45
energy sector	1. Energy awareness	1	90	0,00	0,01	0,16	0,08	0,00	1,88	0,09
energy sector	3. EI of energy prod. equipm.	0,2	18	0,00	0,00	0,03	0,02	0,00	0,38	0,02
energy sector	5. EI energy transportation	2,5	224	0,00	0,02	0,40	0,19	0,01	4,70	0,22
energy sector	4.EI consumers equipm.	0,3	27	0,00	0,00	0,05	0,02	0,00	0,56	0,03
energy sector	6. Red. og energy transmission/distrib. loss	0,6	54	0,00	0,01	0,10	0,05	0,00	1,13	0,05
energy sector	7. Cogeneration	2,7	242	0,00	0,02	0,43	0,21	0,01	5,08	0,24
energy sector	8. Improved energy managem. in buildings	0,3	27	0,00	0,00	0,05	0,02	0,00	0,56	0,03
industry	1. Energy awareness	6,5	489	0,12	0,04	0,50	0,27	0,47	6,10	0,46
industry	2. Updating energy technologies.	2	150	0,04	0,01	0,15	0,08	0,14	1,88	0,14
industry	5. EI energy transportation	0,5	38	0,01	0,00	0,04	0,02	0,04	0,47	0,04
industry	4.EI consumers equipm.	0,7	53	0,01	0,00	0,05	0,03	0,05	0,66	0,05
industry	6. Red. og energy transmission/distrib. loss	0,8	60	0,01	0,00	0,06	0,03	0,06	0,75	0,06
industry	7. Cogeneration	0,1	8	0,00	0,00	0,01	0,00	0,01	0,09	0,01
industry	8. Improved energy managem. in buildings	0,5	38	0,01	0,00	0,04	0,02	0,04	0,47	0,04

industry	9. Thermal insulation	1,5	113	0,03	0,01	0,11	0,06	0,11	1,41	0,11
industry	12. Renewables	0,1	8	0,00	0,00	0,01	0,00	0,01	0,09	0,01
agriculture	1. Energy awareness	2	150	0,03	0,04	0,28	0,08	0,23	0,95	0,47
agriculture	2. Updating energy technologies.	0,5	37	0,01	0,01	0,07	0,02	0,06	0,24	0,12
agriculture	4.EI consumers equipm.	0,5	37	0,01	0,01	0,07	0,02	0,06	0,24	0,12
agriculture	12. Renewables	0,4	30	0,01	0,01	0,06	0,02	0,05	0,19	0,09
TOTAL		**63,7**	**4919,9**	**1,1**	**0,5**	**17,4**	**71,8**	**5,7**	**46,8**	***10,1***
% of total consumption/ emissions in 1992		**7,65**	**7,5**	**9,4**	**7,8**	**10,1**	**12,3**	**10,0**	**5,7**	***9,3***

Appendix 2



NO2-exposure in Budapest
600000
500000
400000
300000
200000
100000
0
People
10-20
20-30
30-40
40-50
50-60
60-70
80-90
Yearly average (µg/m3)



SO2-exposure in Budapest
800000
700000
600000
500000
400000
300000
200000
100000
0
People
0-5
5-10
10-15
15-20
20-25
25-30
30-35
Yearly average (µg/m3)

Appendix 3

Exposure-response functions for health effects. OR or RR = $e^{\beta \cdot C}$, β: regression coefficient, C: concentration level (from Aunan, 1996, unless noted).

Health end-point	**Relative risk model**	**β (95% CI)**	**C unit (averaging time)**	**p_0**[1]
Acute resp. symptoms:				
Acute RS, children	OR	0.003 (0.001 - 0.006)	$\mu g/m^3$ TSP (daily)	0.068 - 0.074
Pseudocroup, children	RR	0.124 (0.064 - 0.185)	log $\mu g/m^3$ TSP (daily)	$8 \cdot 10^{-6}$ - $16 \cdot 10^{-6}$
Acute RS, adults	OR	0.0015 (0.0007 - 0.0022)	$\mu g/m^3$ TSP (daily)	0.026 - 0.030
Asthma symptoms, asthmatic adults[2]	Absolute[3]	0.06 (0.001 - 0.119)	$\mu g/m^3$ $PM_{2.5}$ (daily)	
Chronic resp. symptoms:				
Chronic RS, children	OR	0.025 (0.003 - 0.050)	$\mu g/m^3$ PM_{10} (annual)	0.02 - 0.04
Chronic RS, adults	OR	0.005 (0.004 - 0.007)	$\mu g/m^3$ TSP (annual)	0.04 - 0.06
Crude mortality:				
Total	RR	0.0013 (0.0011 - 0.0015)	$\mu g/m^3$ PM_{10} (daily)	$19 \cdot 10^{-6}$ - $26 \cdot 10^{-6}$
>65 y	RR	0.0018 (0.0014 - 0.0021)	$\mu g/m^3$ PM_{10} (daily)	$12 \cdot 10^{-6}$ - $17 \cdot 10^{-6}$
<65 y	RR	0.0005 (0.0004 - 0.0006)	$\mu g/m^3$ PM_{10} (daily)	$7 \cdot 10^{-6}$ - $9 \cdot 10^{-6}$
Infant mortality (0 -1 y):	OR	0.009 (0.004 - 0.015)	$\mu g/m^3$ PM_{10} (annual)	0.004 - 0.009[4]
Lung cancer:	OR	0.0056 (0.0052 - 0.0060)	$\mu g/m^3$ TSP (long-term)	(0.0003)

[1] Hypothetical zero-concentration prevalence, daily or annual depending on whether the function refers to daily or annual average concentration.

[2] Based on Ostro et al. (1991)

[3] Daily incidence rate = $\beta \cdot \ln C$