REDUCED DAMAGE TO HEALTH AND ENVIRONMENT FROM ENERGY SAVING IN HUNGARY

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1. Summary

The aim of this study was to assess the costs and benefits of the implementation of a specific energy saving program in Hungary. The program constituted the major part of measures to meet Hungary's obligations under the UN Framework Convention on Climate Change, and was primarily designed to reduce the emissions of CO_2. The energy saving expected from the program was approximately 64 PJ/year (7.7% of the total energy consumption).

We have estimated the possible reduced damage to public health, building materials, and agricultural crops that may be obtained from implementing the program. Possible benefits were estimated using a bottom-up methodology where we applied monitoring data and population/recipient data from Hungary and exposure-response functions and valuation estimates (unit prices in terms of damage costs and willingness-to-pay) mainly from US and Western European studies. Our analysis indicates that the main benefit from implementing the program relates to public health, and that reduced prevalence of chronic respiratory diseases and premature mortality are the most important effects. The estimated annual benefit of improved health conditions alone, as estimated by a bottom-up willingness-to-pay approach, is approximately 650 mill. US$ (370 mill. US$ - 1170 mill. US$) and is likely to exceed the investments needed to implement the program. In addition there are significant benefits due to reduced replacement and maintenance costs for building materials (30 - 35 mill. US$ annually in Budapest only). Crop damage from ozone may be large, but a significant improvement in Hungary depends upon concerted actions in several countries.

A marginal benefit of reducing CO_2 emissions has been suggested by others to be in the range of approximately US$ 64 to US$ 164 per ton carbon. Applying these estimates to the Hungarian case study gives an annual benefit from reduction in CO_2 emissions with respect to climate change in the range 86 - 222 million US$, i.e. lower than the ancillary benefits.

Information from the bottom-up assessment was used in a macroeconomic model to carry out an evaluation of the social costs of the energy saving program. The optimal level of abatement was estimated to be about 65 PJ/year, i.e. slightly higher than the original program. A leakage of about 5 PJ was estimated to occur due to improved environment, hence the total energy saving is estimated to be around 60 PJ/year. The leakage is due to increased output in the commodity and service sectors following lower damage, which leads to increased energy use. The value of the Energy Program in the macroeconomic study was estimated at 2.6 cents/kWh, which compares with 3.6 cents/kWh in the bottom-up study based on the willingness to pay. The reason for the discrepancy between this estimate and the bottom-up estimates is that the unit prices applied in bottom-up studies do not reflect the change in marginal cost when the environment improves. This is because the valuation is usually carried out prior to the implementation of the measures, i.e. in a situation when the abatement level is sub-optimal. Since such unit prices are based on constant prices, and the marginal cost of improving the environment is likely to be reduced as measures are implemented, the result may be that the total value of abatement is overestimated in bottom-up studies.

2. Introduction

In environmental policy making there is a continuous need to evaluate alternative measures to improve the quality of the environment. In doing so, the demand for reliable economic assessments increases. To provide reliable assessments, there ought to be consensus about the methodology of valuation. This is the point at which the controversies among economists start, and by use of different methodologies one may end up with widely different estimates of benefits for the same environmental improvement. In a case study in Hungary different approaches were taken to estimate the benefits that may be achieved by improving the environment.

First, a bottom-up study (impact pathway) was made, which included estimated benefits in terms of reduced damage to health, building materials and cereal crops. The physical estimates were evaluated by means of willingness-to-pay and damage cost unit prices, respectively. Secondly, a model for integrating damage costs and demand side assessments from bottom-up studies into a macroeconomic framework was developed, in order to 1) estimate the optimal level of abatement; and 2) to estimate the value of environmental improvements. The present paper is mainly based on Aunan *et al.* (1998) which describes the bottom-up assessment and Aaheim *et al.* (2000), which describes the macroeconomic assessment.

The focus of our study was the benefits that may be achieved from implementing measures that reduce the overall energy consumption, and thereby the general pollution level, in Hungary. The measures are described in the National Energy Efficiency Improvement and Energy Conservation Program (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 are to:

- improve environmental protection;
- reduce the dependency on imports;
- save domestic energy resources;

- postpone 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.

Our scenario was assessed over 5 years and had the baseline economic assumptions that the annual growth rate of GDP was expected to decrease up to 1995. Beyond 1995 the annual growth rate was assumed to increase to 1-2 % per year (in 1995 the growth rate was ca. 1,4% and in 1996 it was 0,8-0,9%). Moreover, it was assumed that 1) the price system of energy carriers should reflect realistic expenditure and the cross financing should be ceased; 2) energy awareness should be developed as a consequence of rise in prices of the energy carriers; and 3) Centralised 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) was allocated to the various sectors and measures, the estimated total investment (capital and operating costs) needed during the implementation period of 5 years was given as an aggregate, the present value being 422 mill. US$ (i.e. the program seems to be highly profitable, as the present value of the saved energy cost would be above 5 bill. US$ if we assume a lifetime of the measures of 15 years). We had some information on the estimated relative needs within some sectors, but it proved difficult to obtain a comprehensive picture of the cost estimate. 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).

Table 1. **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*	5.8 - 7.5

The possible energy saving and estimated emissions reductions are given in Table 1. The emission coefficients for each sector are from Tajthy and co-workers (Tajthy, 1993; Tajthy *et al.*, 1990). In the study it was assumed that the energy saving primarily affects the consumption of fossil fuels, and we also assumed a status quo baseline emission scenario (see Aunan *et al.*, 1998 for a discussion). 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 in view of the aggregated level of the calculations done in the study, 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. To illustrate the approximate share of the concentration level of air pollutants that may be inert to national emission reductions, we used data on the share of deposition of nitrogen and sulfur that are indigenous (EMEP, 1996). Using Budapest as an example, we estimated a likely range for the background level of PM_{10} and a likely range of the contribution to this level from foreign sources. The calculations indicated that around 62% (49%-67%) of the annual average concentration level of PM_{10} in the city would be influenced by national emission reductions. A recent study by Tarrasón and Tsyro (1998) showed that the relative indigenous contribution generally is smaller for $PM_{2.5}$ (the fine fraction of airborne particles) than for sulfur and nitrogen deposition.

3. Reduced health effects as estimated in the bottom-up study

3.1 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 this study apply to specific groups, as elderly or children, they in most cases 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 a subgroup of more susceptible individuals suffers a disproportionate share of the damage.

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 (see Aunan *et al.*, 1998).

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). We found that the NO_2-data could be used. In Budapest NO_2, PM_{10}, and TSP are monitored, in the 18 county capitals NO_2 and TSP are monitored, whereas in the remaining cities only NO_2 is monitored (in addition to SO_2 and dust fallout, which are monitored in all cities). We used the data from Budapest to establish the relations for NO_2 versus PM_{10}, and PM_{10} versus TSP, and used the data from the county capitals to test these relations (see Aunan *et al.*, 1997). We found a PM_{10} /TSP-ratio of around 0.35-0.4, which is lower than what is often reported in studies in USA, where 0.5 - 0.6 is suggested as a conversion factor if no other data are given. 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 M. Krzyzanowski, 1996). We concluded that our method, by estimating the concentration of particles from the NO_2-level, probably at the most underestimates the response by 14% (Aunan *et al.*, 1997). We also concluded that, although the concentration of particles may be underestimated in cities with high levels of particles, 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.

3.1.1 Willingness-to-pay unit prices

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, 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 (chronic diseases and mortality excluded). 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 unit values and estimated benefits are given in Table 2.

If we assume that WTP for health risk prevention takes an increasing share of the budget as income increases, the use of 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 are 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).

Since the end-points in our study are not exactly the same as those valued by the US-EPA (1995), we adjusted some of the estimates, and made some additional assumptions. The WTP for avoiding one case of infant death was 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 (a sensitivity analysis is given in Aunan, 1998). We assumed a mean 5-year survival rate of 20%; this may still be too high (see Scientific American, 1996).

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 could not use the unit price directly. Instead we estimated a modified unit value for what we called "ARS-restricted", taking the daily wage multiplied by the WTP/COI ratio of 2 (our estimate became around 3 times higher than the RAD-value in US-EPA (1995). In addition to this, we 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 used 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.

Table 2. **Unit values (willingness to pay) for health impacts (1994 US$), and estimated annual benefit from implementation of the energy saving program 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>65y	3.4	1.9	6.8	0.548	0.306	1.097	261.2	103.5	663.4
Deaths≤65y	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	121.5	1.8	1042.4
CRS-Adult	0.24	0.14	0.38	0.039	0.023	0.061	143.0	58.2	347.1
TOTAL							648	192	2549

Notes: [1] Mill. US$ unless noted; [2] US$.

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 day losses for one parent. This COI-estimate was multiplied with a WTP/COI ratio of 2 (US-EPA, 1995). The severity of the chronic bronchitis cases in the basis study (Abbey *et al.*, 1993 and 1995) used by the EPA to estimate a unit value 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 (Abbey *et al.*, 1993) are approximately the same. 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). The adjusted unit price used to estimate the *annual* benefit of reduced chronic bronchitis rendered in Table 2 is calculated by assuming a duration of 5 years and a discount rate of 6%. A sensitivity analysis showed that the total health benefit is rather sensitive to assumptions about severity and duration of cases of chronic bronchitis (see Aunan, 1998). If we, for instance, assume 10 years instead of 5 years, the total benefit estimate falls by nearly 20%.

3.2 Bottom-up estimates of economic benefit of reduced health damage

The total health benefit estimated to be achieved by implementation of the energy saving program is given in Table 3. The uncertainty intervals 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 (if this value is used), and the uncertainty in the conversion factor between various particle measure (if conversion is needed). The procedure for estimating the reduced health damage is given in Aunan *et al.* (1998). The aggregated uncertainty in the total health benefit estimate arising from uncertainties in important input parameters and variables is, however, better represented by performing Monte Carlo simulation. The overall probability distribution is shown in Table 3.

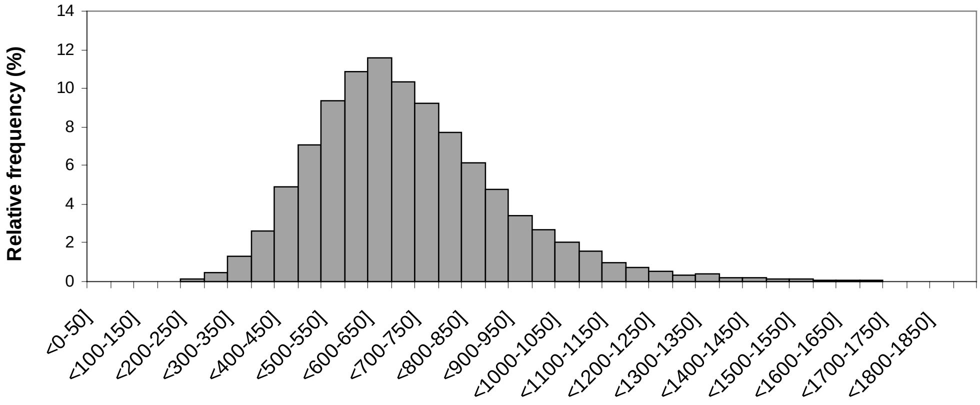
Table 3. **Estimated annual benefits and costs (mill. US$) of implementing the energy saving program NEEIECP**

(Results from the bottom-up study.
Excess mortality in people > 65 years of age is included in the health effects estimate)

	Best estimate	Range
Health effects	648	370 - 1168
Materials[1]	105	60 - 150
Vegetation	1.5	0.9 - 2.2
Climate[2]	154	86-222

1. The uncertainties are subjectively estimated to be about 40%.
2. Most estimates of the marginal CO_2 emission cost lie in the range 64 US$ to 164 US$ per tC. Our ‘best estimate’ is obtained using a value close to the upper limit of this interval.

Figure 1. **Probability distribution for the total health benefit of implementing the energy saving program in Hungary, as it was estimated in a bottom-up study. 10,000 trials**



The ancillary benefit per ton C, in terms of reduced health damage as it was estimated in the bottom-up study (about 482 USD/tonC), is high compared to other studies (see e.g. Krupnick and Burtraw, 1996 and Ekins, 1996). One important reason for this is that we used a separate, steeper function for mortality in people >65 years of age. This gives a considerably higher number of deaths than other studies for this group (see Table 4 in Aunan *et al.* 1997). At the same time the economic unit value for deaths (VSL) applied for this age group is quite high; 75% of the VSL for people < 65 y of age. Also, we have included a wider range of health end-points than many other studies. Moreover, the concentration level in many of the Hungarian cities is considerably higher than at least in the US and Western Europe, especially when it comes to particles. The rough approximation of assuming that a given percent overall reduction of the emissions of an air pollutant gives the same percentage reduction in the average concentration level in the cities could also have lead to an overestimation of the health benefit.

3.3 Reduced material damage

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 (see Kucera and Fitz, 1995).

We used the results from a study of the economic loss due to damage of materials in Budapest (Kruse, 1995) to assess the effect of the implementation of NEEIECP in the city. The original study used statistics of building mass and materials in the city, together with results from studies in other European cities, to assess the damage to materials. The methodology and damage functions given in Kucera and Fitz (1995) were applied, which involved estimating the reduced maintenance and replacement costs obtained from reducing the SO_2-concentration level with certain steps (see details in Aunan *et al.*, 1998).

Using 1990 as the baseline year Kruse (1995) estimated the annual saving in total corrosion costs to be about US$ 50/inhabitant, if average SO_2 levels were reduced to less than 20 $\mu g/m^3$ in all regions of Budapest. This implies a total annual saving of US$ 100 mill. In 1990 an area representing 10-20% of the city area (in five districts in Budapest) had annual mean SO_2-concentration above 20 $\mu g/m^3$. We estimated that a flat 6% reduction, which is the estimated average SO_2-reduction resulting from implementation of NEEIECP, should reduce the area having a SO_2-level above 20 $\mu g/m^3$ with 20-25%. Taking into account the building density in the various districts, we arrived at a reduced annual cost in the range 30-35 mill US$. An extrapolation of this figure to urban Hungary, using the population, gives the estimate in Table 3.

4. Alternative approaches to evaluate the benefit of an improved environment

Cost-benefit analysis of specified measures to improve the environment has traditionally been based on bottom-up assessments. As mentioned, there are alternative approaches to choose among, such as willingness to pay or cost of illness. These approaches usually turn out with widely different estimates. In this section we discuss why this is so, and suggest how both approaches may be used to do a macroeconomic analysis in order to obtain better estimates.

4.1 The reference point for alternative estimates of value of the environment

The alternative approaches of valuation are illustrated in figure x. The thick *MC*-curve represents the marginal cost of energy savings, calculated as the cost per saved PJ of energy, and ranked according to the cost per unit of saved energy. The Energy program saves *x(1)* units of energy, with a marginal cost equal to *p(1)*. This corresponds to the unit cost of the most expensive measure, insulation and renewable energy. Without considering the environmental benefits, the usual cost benefit criterion is whether the price of energy exceeds the unit cost of the whole program, that is, whether the reduction of the electricity bill exceeds the dark shaded area. Note, however, that this is not strictly correct. If the energy price is lower than *p(1)*, this criteria could be satisfied, but some of the more expensive measures should not have been implemented.

One way to take environmental benefits into account is to include indirect environmental benefits from associated reduced emissions, and subtract them from the marginal costs. This leads to a negative shift in the marginal cost curve, from *MC* to the marginal social cost curve, *MSC.* Hence, the entire energy saving program might be socially beneficial, even if the alternative price of energy is lower than *p(1).* In the example displayed here, inclusion of environmental benefits turns the marginal social cost of the program negative, even if the price of energy is zero. Below, we assume that the energy price is equal to zero.

An alternative approach to include environmental benefits is to consider the willingness-to-pay for improved environmental quality, for example by a questionnaire to a sample of people. An estimate of the willingness to pay (*p(WTP)*) determines the point on the demand curve at which no energy saving has taken place, i.e. at $x = 0$. The bottom-up cost benefit criterion could be, either that *p(WTP)* should exceed *p(1)* or that the light shaded area should exceed the dark shaded area. Neither are perfect, because the willingness to pay and the marginal cost refer to different quantities of energy conservation, *x(0)* and *x(1)*, respectively. For large changes, the willingness to pay will decline as the environmental quality improves. Furthermore, the willingness to pay should be compared with *MSC*, to take account for reduction in damages as well.

The two approaches may yield widely different results, but none of them are correct. If energy savings actually were carried out, the willingness to pay for less pollution would decrease. Moreover, if the Energy Program is a no-regret option, it is clearly beneficial to save more energy than *x(1)*. Implementation of new energy saving measures would establish a new equilibrium, where the marginal cost equals the marginal willingness to pay, i.e. *p(2)*. This is the only point where the value of the environmental quality is defined, and the marginal social cost of the energy program is equal to the marginal willingness to pay. The amount of energy saving is then *x(2).*

The critical assumptions underlying a standard bottom-up approach are, first, that average costs and benefits are interpreted as marginal costs and benefits. Second, that the use of either willingness to pay or damage cost as a proxy for the value of the environment presumes equilibrium, although the aim of the same studies frequently is to show disequilibrium. The main problem is in most cases not the use of a bottom-up approach, but rather that market effects are ignored. One solution could therefore be to carry out a partial analysis of the 'environmental market', still based on the bottom-up information, but where the marginal cost of each measure were compared with the marginal willingness to pay.

An alternative is to implement the environmental market, including the energy saving program, in a macroeconomic model. This also involves problems, because it is not possible to account for all the available information about specific measures provided by the bottom-up study. The advantage is that the relationship between the resource use of energy saving, and the economic impacts of cleaner air across the whole economy can be fully accounted for.

The macroeconomic model applied here is described in Aaheim *et al.* (2000). It was made as simple as possible in order to highlight the valuation issue. One production sector produces all commodities and services, except health services. The production sector is polluting, and delivers its output to the health sector, to households and to 'produce' abatement measures. In addition, it uses some of its output as input in own sector. Production is affected by air quality in terms of crop losses and material damage.

Households buy products from the commodity and service sector, and health services. The demand for health services was calibrated by the willingness to pay retrieved from surveys in other countries, adjusted for income level and level of air pollution. Health services are produced by means of labour and input from the commodity and service sector. The activity in the health sector is given by assumption in the case where no energy saving is carried out. To analyse the value of the Energy program, the activity in the health sector is determined by the demand for health services in households.

4.2 Comparing the results from the bottom-up and top-down analysis

The estimated annual benefit of the energy saving program obtained by the different approaches differ considerably, as shown in Table 4 (from Aaheim *et al.*, 2000). The bottom-up WTP estimate given in Table 4 is based on the assumption that the air pollution level in Hungary is approximately three times higher than in the US, and does not include excess mortality in people above 65 years of age (see Aaheim *et al.*, 2000 for details). The benefit estimated in the top-down analysis is, as expected, considerably lower than in the bottom-up analysis applying WTP estimates. Moreover, the estimate obtained by applying damage cost estimates typically is lower than when WTP is used, because the damage costs, such as COI, comprise market-values only. If damage cost estimates covered all damage, including the cost of the measures needed to reduce emissions, and the respondents behind a WTP estimate were 'ideal', e.g. disposing full information, one could maintain that market failure is the only reason why WTP diverged from the full marginal social cost of environmental measures. In practice, however, this is not likely to be the case. Assessment of the WTP is hampered by large difficulties (see Navrud and Pruckner, 1997). Moreover, it is difficult to make a full assessment of the damage costs, for example because necessary data are unavailable or incomplete. Symptomatically, much of the difference between the B-U estimates in the study by Aaheim *et al.* (2000) is due to the diverging estimates for chronic diseases an premature mortality, where welfare losses are likely to be more prominent than for many other end-points. Both from a practical and a theoretical point of view, it therefore seems unwarranted to apply a general WTP/COI ratio in cost-benefit analyses, as was done in the bottom-up study (Aunan *et al.*, 1998).

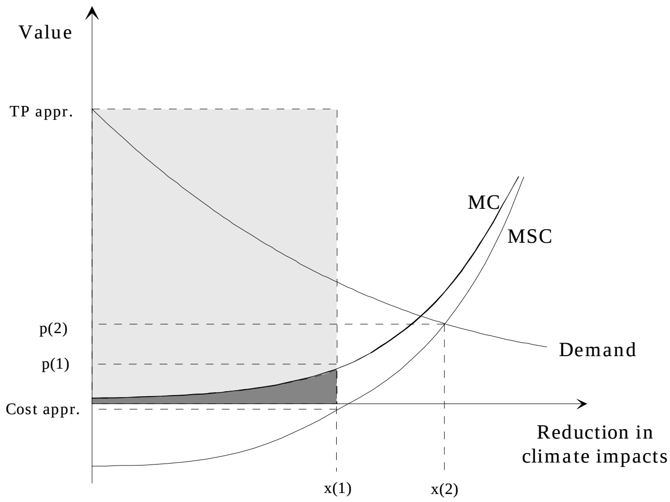
Table 4. **Alternative assessments of energy saving potential and environmental benefits of the Energy Saving Program**

Method	Energy conservation (PJ)		Total value of ancillary benefits	Marginal value	
	Energy Program	Reduced energy use	Mill USD	Mill USD/PJ	Cents/ kWh
Bottom-up					
Damage cost	63.7	63.7	141.5	2.2	0.8
WTP	63.7	63.7	645.0	10.1	3.6
Top-down	64.5	60.0	43.5	7.3	2.6

Note:
The bottom-up damage cost estimate includes health, building materials, and agricultural crops. The bottom-up WTP estimate includes only health (see text for details). Excess mortality in people > 65 y of age is excluded in all three assessments.

The estimated market equilibrium point implies a recommendation that energy saving measures corresponding to about 65 PJ, *x(2)* in Figure 2, should be implemented (i.e. very close to the original program of 63.7 PJ). The defined unit value of *environmental quality*, i.e. the price of energy saving, in Figure 1, was about 7.3 mill. US$/PJ. The corresponding total abatement cost, i.e. the integral under the cost curve in Figure 2 was 43.5 mill. US$. These estimates are, however, very sensitive to the various assumptions made (see Aaheim *et al.*, 2000).

Figure 2. **Alternative approaches to valuation of environmental change**



5. Conclusions

Estimates of the value of environmental quality may depend strongly on the approach. In particular, estimates based on bottom-up studies may differ considerably according to the method. However, one should not consider alternative bottom-up approaches as ‘competing’, but rather as means to provide supplementary information for an assessment of the value of the environment. In principle, valuation ought to be assessed in a macroeconomic context, but bottom-up estimates may be appropriate when considering small changes. It may be tempting to argue that ‘small’ could be considered as changes that do not affect macroeconomic variables significantly. This may be an insufficient requirement. The results indicate that although the over-all macroeconomic effects of the Energy Program are negligible, the macroeconomic effects may be important to the values attached to the Energy Program, such as the marginal value of emission cuts and the amount of leakage due to a better environment, etc. In addition, a macro economic study provides additional information about the allocation of the environmental benefits by distinguishing between consumption and health status.

A disadvantage with a top-down approach is that a specification of measures must be expressed in general terms. For instance, the measures included in the Energy Program were expressed in terms of a cost function, which is based on a number of assumptions. The results turned out to be very sensitive to the assumptions about the shape of the cost curve for abatement measures, and to the willingness-to-pay estimate. Usually, cost curves can be established with more reliable information than was available for this study, where this proved to be difficult. Assessments of the demand for environmental qualities, such as willingness-to-pay estimates, are much more problematic. This applies in particular when based on studies in other countries, as in the present study.

The Energy Program was originally presented as a mean to reduce greenhouse gas emissions in order to make Hungary keep track with the expected commitments in an international treaty on greenhouse gas emissions. Although the value estimates of this study were based on very rudimentary information, the results indicate that the local effects on pollution probably make at least a part of the Energy Program a ‘no-regret’ option. This may be the case for many measures planned to reduce emissions of greenhouse gases, but local, or ‘ancillary’, effects of climate measures are seldom taken into account when assessing the costs of climate policy. The results here indicate that this may be a serious deficiency.

Many methodological challenges have not been discussed in this paper. One problem, related to the implementation of a cost function for energy saving measures in a macroeconomic model, is to rank the measures appropriately. In this study, the measures that constitute the Energy Program were ranked according to the unit costs per saved amount of energy. The demand for emission cuts (or energy saving) is, however, related to the demand for health services. In general, the relationship between energy saving and the effects on health varies between measures, for example because the population exposed to pollutants differs in different areas. How to do a ranking that appropriately takes these factors into account is a subject for future research.

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