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Social Benefits of Energy Conservation in Hungary

An examination of alternative methods of evaluation

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PREFACE

This study is a follow-up of a feasibility study of environmental policy making in Hungary that was worked out in co-operation with Hungarian researchers in 1995 and 1996. We are greatly indebted to Gyula Bándi, János Szlávik, Edith Tajthy, Tihamér Tajthy and, in particular, Gyõrgy Pátzay for their contributions to earlier reports. Without their efforts, this study would have been impossible.

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ABSTRACT

This study aims at comparing alternative methods for a valuation of the social benefits of less air pollution. Estimates of such benefits can be based either on the economic consequences of less damage associated with pollution, or on estimates of the willingness to pay for less pollution. The damage cost approach is based on relatively reliable data, but disregards direct welfare gains of a better environment. Such gains can be assessed by the willingness to pay approach, which, however, is problematic for methodological reasons. In this study, we assume that it is possible to estimate the willingness to pay in a meaningful way.

Using a program for conservation of 64 PJ fossil energy in Hungary as a case, the damage cost approach estimates the benefits to be nearly 75 mill. USD per year, dividing into approximately 41 mill. USD for health, and 33 mill. USD in material damage. The willingness to pay approach, on the other hand, yielded a value of 1.9 bill. USD. Most of this difference is due to the different estimates for chronic distresses and early death. Ignoring methodological problems, the main reason why the two approaches diverge is that both refer to an initial situation, *before* the reduction in air pollution has taken place. The 'correct' estimate would be to calculate value and quantity of reduced emissions in equilibrium, *after* the Energy Program has been implemented. To do this, a very simple and aggregated macroeconomic model was applied.

The equilibrium point on the Energy Program curve was estimated to be 70 PJ, or approximately 10 percent above the reported amount of energy conservation in the program. The marginal cost of the program is 7.5 mill. USD/PJ, while the marginal cost in equilibrium (the price of energy conservation) was estimated to 27 mill. USD/PJ. With more moderately increasing abatement cost, the price of energy conservation in equilibrium was 21 mill. USD/PJ, yielding a total cost of 1.9 bill. USD for energy conservation. This is the same as the willingness to pay for the program in the bottom-up approach. The corresponding energy saving was 91.2 PJ. If the willingness to pay is reduced to ¼, the price of energy conservation dropped to 5.1 mill. USD/PJ. This means that a small part of the Program should be discarded.

The results of the different approaches should not be regarded as alternative answers to the same question. Rather, the damage cost and the willingness to pay assessments can be considered as necessary components of a macroeconomic study of environmental policy. To carry out a study relevant for policy making, however, requires better data, especially on economic conditions.

1 INTRODUCTION

In their communication to the UN Framework Convention on Climate Change, the Hungarian Ministry of Industry and Trade (1994) presents a program to improve the energy efficiency of the economy. If implemented, it is claimed that the program will contribute to a reduction in energy consumption in Hungary of approximately 6 percent. OECD/IEA (1995) has criticised the communication for not taking economic aspects of the program sufficiently into account. Their main objection is that the government does not consider the possibility for funding the program. The estimated, total cost of the full program is reported to be 422 mill. USD. This figure is not well documented according to the IEA. Moreover, it is an open question how the total cost is distributed among the different measures included in the proposal. With the prevailing problems of the Hungarian economy in mind, it is unlikely that implementation of the program will be given a high priority.

The problems of assigning reliable economic estimates to policy proposals are common, and they are especially pronounced in countries in transition, such as Hungary, which are subject to substantial market imperfections. This applies especially in the energy markets, which are subject to strong regulations in many countries. Whether it pays to invest in measures to save energy is an even more difficult question than to assess the costs, since the social benefits of reduced pollution levels will have to be assessed. Since these benefits may contribute significantly to the financing of the energy saving measures, it is useful to examine what type of economic data is required to make such assessments.

This paper discusses alternative methods for economic evaluation of environmental policy measures. Our purpose is partly to show what kind of economic information is required, and how sensitive the social benefits of proposed measures are to economic assumptions. Moreover, we show how political priorities over environmental issues affect the priority over measures. The energy saving program of Hungary is used as an example, but the main purpose is not to advice Hungarian authorities about environmental policy. Rather, the Hungarian case is considered as a typical example of the availability of data, and therefore a realistic point of departure for environmental policy analysis for economies in transition.

Available information on the energy saving potential and associated effects on emissions to air is often more complete than on economic figures. Moreover, the availability of data of relevance for an assessment of the effects of emission reductions is relatively good in Hungary. By means of such data, Aunan *et al.* (1997) estimated the effects of the energy saving program on humans, materials and crops. Even though the uncertainties are large, the decision-makers have access to rather detailed information about the effects of the measures, but are left with highly insufficient information about resources needed to implement them. This makes it difficult to compare alternatives to the energy program and to rank the measures within the program. It turns out, however, that a limited augmentation of economic data would significantly enhance the ability to make a social evaluation of the program.

The paper consists of two parts. The first part, section 2, presents a bottom-up approach to the evaluation of the measures included in the Hungarian energy saving program. It contains a brief presentation of the program, and discusses additional information necessary to make economic evaluations. Based on some reasonable, although rather arbitrary, assumptions, we present alternative figures of relevance for policy evaluation and comparisons of energy saving measures. We conclude this part by pointing out some limitations of the bottom-up approach. The second part, section 3, presents a top-down assessment of the energy program, based on a highly aggregated model of economy at large. The aim of this presentation is to demonstrate how detailed micro-based information may be utilised in a macroeconomic model, and to show how the priority of alternatives may change when considered in a broader setting. The paper concludes by pointing at the requirements for data for more realistic analyses, and to the need for comprehensive, integrated analyses in general.

2 A BOTTOM-UP APPROACH

2.1 The energy efficiency and conservation program

The energy efficiency and conservation program, henceforth called the Energy Program, lists 13 measures distributed over 6 sectors of the Hungarian economy (Pálvölgyi and Faragó, 1994). Total, annual saving of energy amounts to 63.7 PJ. This is about 6 percent of energy consumption in Hungary in 1993. The measures included in the program range from energy awareness, which is expected to constitute half of the energy-saving potential, to investments in new technologies. The potential is considered to be highest in the residential sector, but substantial reductions may be attained also in transport and in the industry sector. The energy sector has a great energy saving potential compared with the present energy use, while the potential in agriculture is considered to be limited.

The economic information is basically limited to the estimate of total costs, which is 422 mill. USD, although indications of the relative cost in some of the sectors have been given. In order to evaluate the program, the costs of each measure have to be specified. To establish cost figures for the measures in the program, the 13 measures were aggregated into 6 groups. According to a rather approximate information, the total costs of the program were then allocated to each of these 6 groups. Table 1 shows how the measures were grouped together.

The figures are subject to several sources of bias. Firstly, the total cost of the program has been questioned. Secondly, it is only vaguely indicated how the total costs are divided between the measures. Thirdly, we have assumed that the cost per unit of saved energy is the same for all measures contained in each of the six aggregates. This is certainly not the case, but may be acceptable as a first approximation. Finally, the cost of each of the 13 measures originally proposed is probably not invariant over economic sectors, as assumed here.

The unit cost of a measure depends on its time of duration. For some of the measures this depends on the depreciation of the capital equipment. Other measures, such as energy awareness, may aim first and foremost at advancing an ongoing process. Then, the time of duration may be considered as a question of how many years it takes before the measure is implemented if the program is not launched. For these ‘soft’ measures, such as management, we have assumed that this will take 10 years, except in the case of awareness, which we assume will be enhanced without the program in 7 years. For measures requiring new technology, we have assumed 15 years duration, reflecting the assumed rate of depreciation. An exception is insulation and renewable technologies, which are assumed to last for 25 years.

Table 1 Aggregates of the proposed energy saving measures

Measure	Measures in energy program	Annual saving of energy (PJ)	Assumed duration of measure (years)
Awareness	• Enhance energy awareness	34.5	7
Technology update	• Update energy technology in industry • Update energy technology in agriculture • Efficiency improvement in energy production • Efficiency improvement in household's utilities	7.2	15
Reduce loss	• Improve efficiency in energy sector • Reduce loss in transmission and distribution • Co-generation of electricity	7.2	15
Energy management	• Improve energy management in buildings	2.8	10
Insulation and renewable energy	• Improve thermal insulation • Use of renewable energy	2.5	25
Transport	• Improve co-operation in public transport • Reduce energy use in vehicles	9.5	10

It is unclear whether the reported total cost of the Energy Program represents a discounted value of present and future costs, and there is no information about the division into investments and operating costs. In the sequel, we assume that the 422 mill. USD are investments only, and that there are no operating costs, alternatively that operating costs are included by their present value. Information about the choice of discount rate is not provided. The discount rate affects both the unit costs and the annual costs of the total 422 mill. USD cost of the program. The benefits, either in terms of saved quantity of energy or in terms of environmental benefits, are measured in annual terms. The discount rate will therefore affect the social profitability of the program. For instance, a 5 percent discount rate means that the annual cost of the program amounts to 53 mill. USD. A 15 percent discount rate enhances the annual cost to 81 mill. USD. In the sequel, we apply a 10 percent discount rate. This gives an annual cost at 66.4 mill USD.

2.2 Ranking according to unit cost

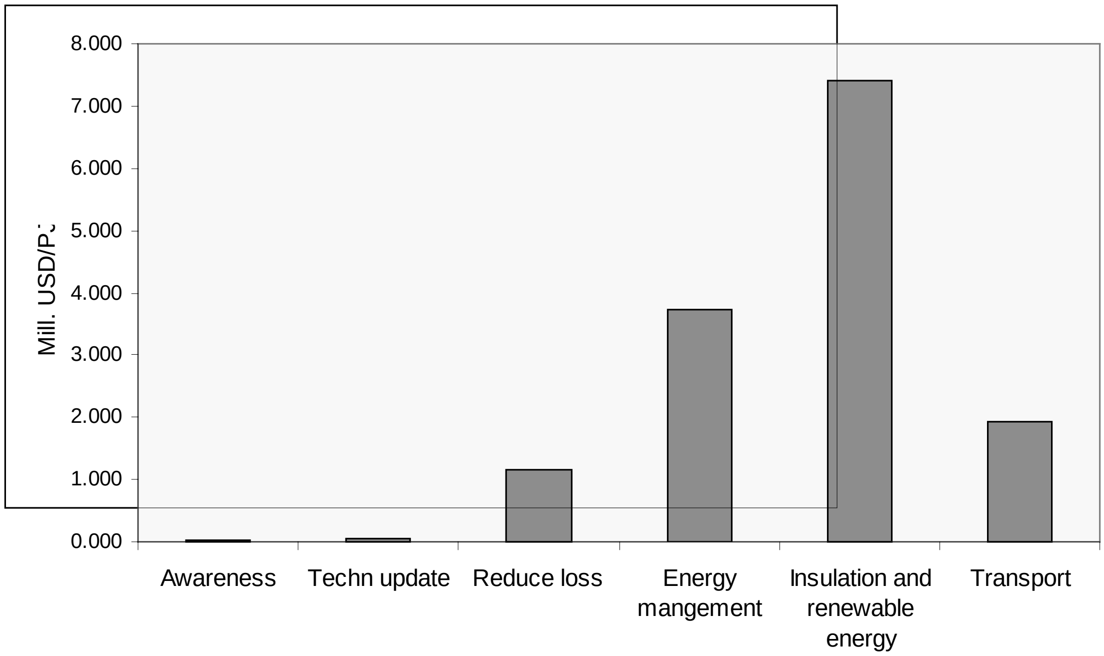
In order to compare the alternatives according to their costs, we define the unit cost, c_i, as the constant cost per unit accrued on the measure over its lifetime. The unit may be the amount of energy saved per year, reductions in the emissions of a particular gas per year etc. Now,

$$c_i = \frac{\sum_{t=0}^{T} \frac{C_t}{(1+r)^t}}{\sum_{t=0}^{T} \frac{x_t}{(1+r)^t}}$$

C_t is total costs at t (in our example 422 at t = 0 and 0 all other years), x_t is reduction in energy consumption or emissions at t, r is the discount rate and T is the lifetime of the measure.

Figure 1 shows the cost per saved kWh energy for each measure, based on the aforementioned assumptions. The unit cost differs substantially between the measures. Hence, from an economic point of view one ought to consider whether the full program should be implemented, or if parts of it should be left out. Recall, however, that the costs are allocated on the different measures on a basis of vague indications. Hence, the figures should be used mainly for illustrative purposes.

Figure 1 Unit costs for measures in the energy program. Mill. USD per PJ



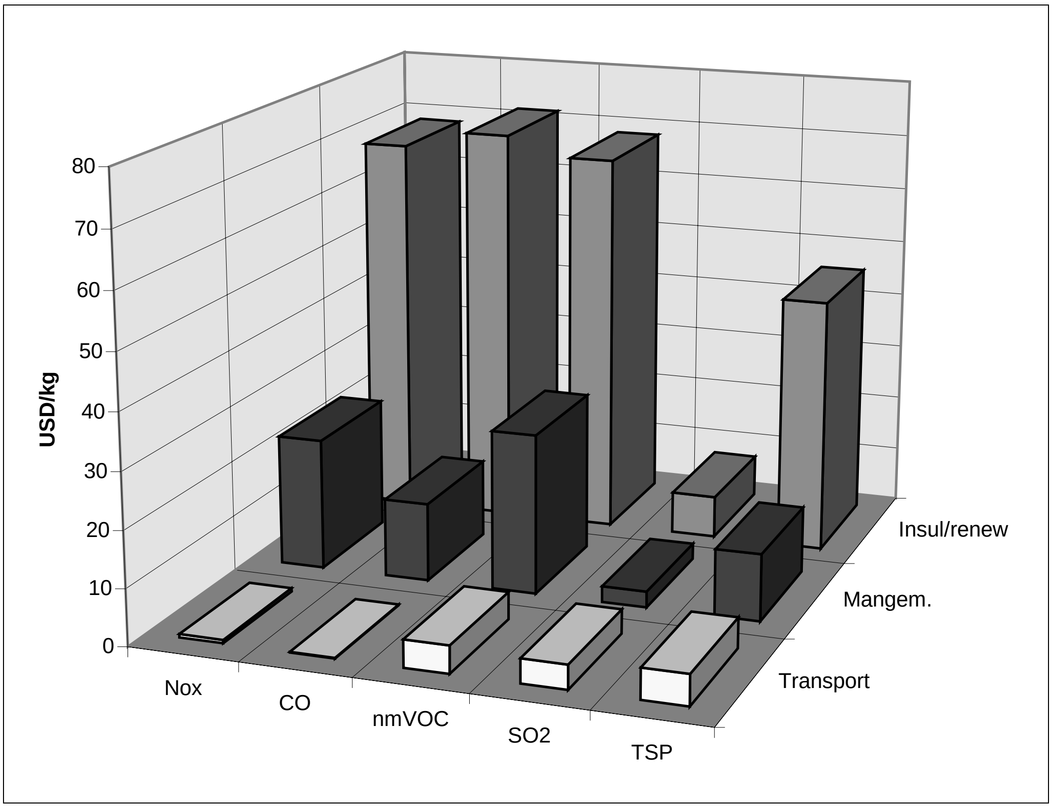
If the unit cost of energy saving is lower than the alternative price of energy, the energy saving measure is socially beneficial, even without including environmental benefits. The overall unit cost of the program is approximately 0.37 US cents per kWh. Hence, the program should be implemented if the alternative price of energy is less than 0.37 cents/kWh in the sectors. With lower alternative prices, additional benefits will have to be documented if the program is to be launched, or parts of the program could be discarded in order to reduce the costs. For instance, if insulation and renewables are taken out of the program, the overall unit cost reduces to approximately 0.24 cents/kWh.

The 'true' alternative price of energy in Hungary is probably much higher than 0.37 cents/kWh. Except for insulation/renewable energy and management in buildings, all the measures save energy at a cost lower than 1 cent/kWh. As a comparison, the electricity price in most Western European countries exceeds 10 cents/kWh. On the other hand, most of the energy saving potential in the program applies for coal, which is less costly than oil in most countries, down to ¼ in some EU-countries (Boug and Brubakk, 1996). Moreover, the energy saving potential for each measure must be regarded as uncertain. In particular it is unclear what enhanced energy awareness implies, how it could be implemented, and what the possibility is for implementing the whole potential or just parts of it.

If the unit cost of a measure exceeds the alternative price of energy, environmental impacts of energy saving may contribute to turn the measure socially beneficial. Then, a valuation of environmental benefits is required. However, in some cases the knowledge about the value of environmental improvements is poor, and sometimes policy aims at achieving specified targets about emission reductions. In such cases, estimates of the costs per unit of emission reduction provide useful information.

Assume for example, that the alternative price of energy is 0.5 cents per kWh. This means that the ‘awareness’-, ‘technology update’- and ‘reduce loss’-measures are socially beneficial without consideration of the environmental effects. The remaining three measures require environmental benefits at least as high as the cost not covered by the alternative price of energy. How much depends on the gas used as basis. Figures 2 and 3 show the excess costs per unit reduction in the emission of the different gases for each of these measures. The excess unit costs correspond to the minimum required environmental benefit of reducing the emissions of *one* gas if the respective measure is to be regarded as socially beneficial. The effect of the Energy Program on emissions is taken from Aunan *et al.* (1997).

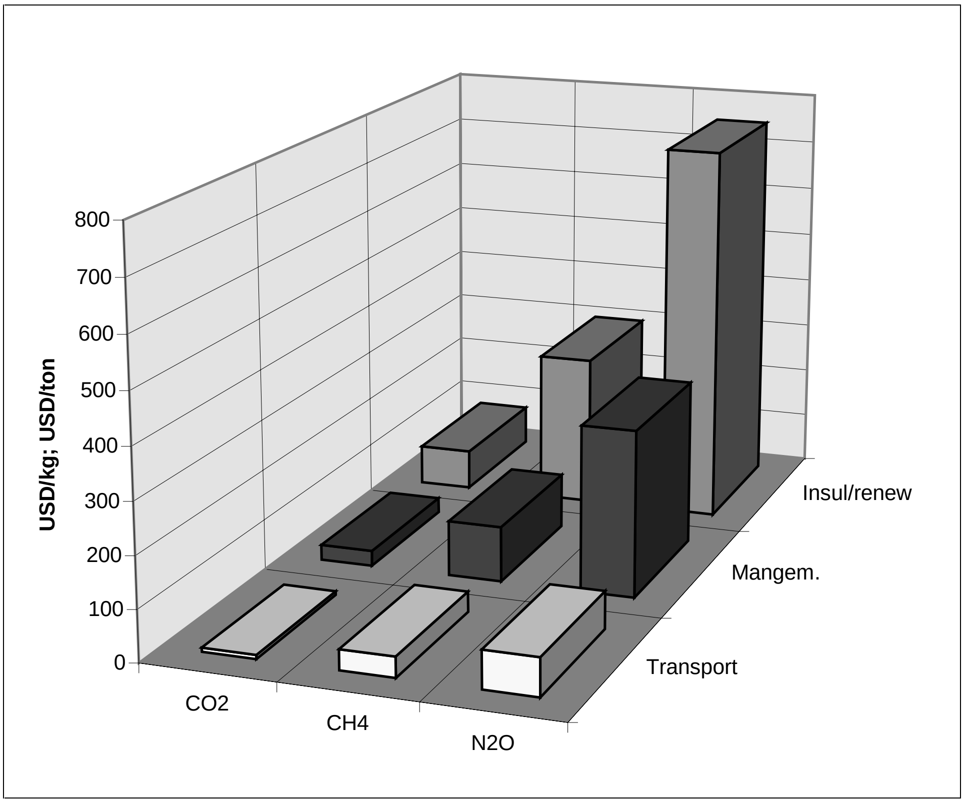
Figure 2 Excess unit costs for reductions of pollutants by three measures. USD/kg



As expected, the most expensive measures in terms of energy saving are also the most expensive when it comes to the excess cost of reducing the emissions. For the pollutants, insulation and use of renewable energy is the most expensive measure, and should therefore be ranked after the two others. Measures in the transport sector have the lowest excess cost for reduction of all the gases, except for SO_2. The excess unit cost of reducing emissions of NO_x and CO are negligible, 0.59 and 0.12 USD/kg, compared with the unit cost of the other two measures, which are between 10 and 100 times higher. For nmVOC (non-methane volatile organic compounds), the excess unit cost of the transport measures is approximately 1/5 and 1/10 of those for management and insulation/renewable energy measures, respectively.

If the only aim of implementing the measures is to reduce the emissions of SO_2, for instance because of an international treaty, the ranking of measures changes slightly. A better energy management in buildings yields more reductions per dollar spent than for the other two measures. With an alternative price of energy at 0.5 cents per kWh, the required environmental benefit for reductions in SO_2-emissions ranges from 2.80 USD per kg for management to 7.50 USD per kg for insulation and enhanced use of renewable energy.

Figure 3 Excess unit costs for reduction in emissions of climate gases for three measures. USD/kg for CH_4 and N_2O. USD/ton for CO_2.



In the case of the climate gases, the ranking is indisputable. Measures in the transport sector are the most favourable of the three. Better energy management is preferred to insulation and use of renewable energy. The excess unit cost of implementing measures to insulation and renewable energy is approximately 10 times higher than measures in the transport sector and twice as high as the management measures for all the climate gases.

The costs displayed in figures 2 and 3 are useful in assessments of the marginal cost of reducing emissions of single gases. This is often focused in international negotiations about transboundary environmental problems, such as acid rain, stratospheric ozone and climate change. Most economic analyses of climate change focus entirely on CO_2-emissions, and early analyses of acid rain were concentrated on SO_2-emissions. For Hungary, it may be useful to assess the unit costs of reducing the emissions of CO_2, not only because it is vital to know the costs of possible future commitments imposed on the country. Possibilities for trading quotas or opportunities for joint implementation make information of the unit cost of CO_2-emission reductions vital. The excess cost of CO_2-emission reductions then gives the minimum price Hungary should require in order to accomplish the measures. Hence, the offer from Hungary's part would be about 7.40 USD/ton CO_2 for implementing the transport measure, 30 USD/ton for management measures and 79 USD/ton for better insulation and use of renewable energy.

2.3 Assessment of benefits

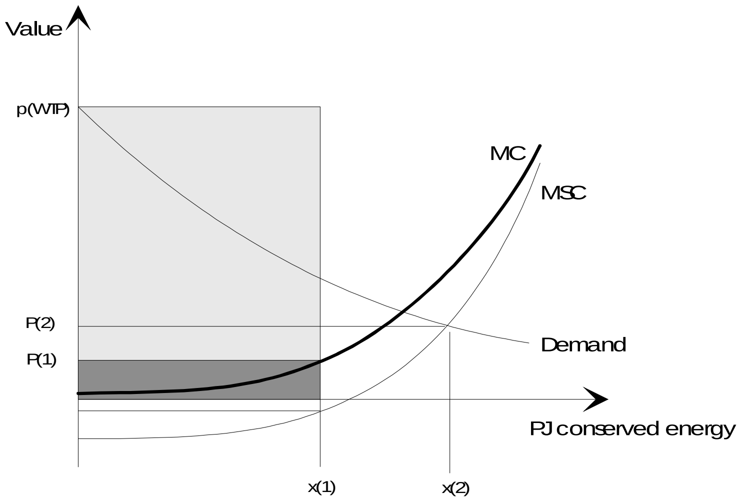
There are two approaches to the estimation of the value of emission reductions. One is to estimate the damage cost. This is an assessment of the economic consequences of the emissions. The valuation of the damage is usually based on observable costs. This implies, for instance, that the damages of pollution are estimated by the cost of restoring the damage, or by the direct and indirect costs of the health effects. Such estimates can be made in various ways, see e.g. UNSO (1992), and the choice of method is usually based on a pragmatic approach, where the availability of data plays an important role. The second approach is to estimate the willingness to pay for a reduction in emissions, or the compensation needed to accept the current level of emissions. This is clearly more demanding than to use the former approach. It is also problematic both for methodological reasons and because it is difficult to make such assessments representative for the population. On the other hand, information about the willingness to pay may in some cases be inevitable in cost-benefit analysis. In this section, we first discuss the difference between the damage-cost approach and the willingness-to-pay approach. Then, the alternative estimates for the Energy Program are examined.

2.3.1 Environmental benefits: Damage costs or willingness to pay?

In a strictly economic sense, the value of a cleaner environment is defined only when the marginal willingness to pay equals the marginal cost for environmental improvements, that is, when the demand for an improvement equals the supply. This is where market equilibrium is established, and the market price is an expression for the value. The need for estimating values occurs for certain goods, not traded in markets, for which prices cannot be observed. In a bottom-up approach, which is based on observed prices, one has to ask what the price would have been if the good was traded. Since observed prices refer to the values prior to the implementation of the measures, the two approaches may give widely different results. There is, however, no general answer as to whether estimates based on marginal costs are better or worse than estimates based on the willingness to pay.

Figure 4 illustrates the difference between the alternative approaches of valuation used in this study. The thick *MC*-curve represents the marginal cost of energy savings, calculated as the cost per saved PJ of energy (see figure 1). Assume that the reduction in energy consumption from the Energy program equals *x(1)*. The marginal cost of the program is *p(1)*, which correspond to the unit cost of the most expensive measure, insulation and renewable energy. The cost-benefit criterion of the program is that the marginal cost of the program is lower than the alternative price of energy. One might find that only a part of the project is socially beneficial. In a macroeconomic context, therefore, the whole program is sometimes regarded as marginal. Then, the test is whether the alternative price of energy exceeds the unit cost of the whole program.

Figure 4 Alternative approaches to valuation of environmental change



The damage-cost approach is based on the assumption that the indirect environmental benefits from associated reduced emissions should be subtracted from the marginal costs. This leads to a negative shift in the marginal cost curve, from *MC* to the marginal social cost curve, *MSC*. Hence, energy saving might be socially beneficial, even if the alternative price of energy is lower than *p(1)*, that is, if the total cost savings of less energy use are smaller than the dark area in figure 4. In the example displayed here, inclusion of environmental benefits turns the marginal social cost of the program negative, a so-called no-regret option.

Assume that the alternative price of energy is zero. By the willingness-to-pay approach, one attempts to examine the demand for environmental improvements, or for energy conservation. An estimate of the willingness to pay (*p(WTP)*) determines a point on the demand curve where no energy saving has taken place, i.e. at $x = 0$. Usually, it is required that *p(WTP)* should exceed *p(1)*, if the improvement is to be considered socially beneficial. This gives an additional total benefit equal to the light grey area in figure 4. This is not a perfect criterion, since the willingness to pay and the marginal cost refer to different quantities of energy conservation, *x(0)* and *x(1)*, respectively. Hence, for large changes, the willingness to pay may exaggerate the benefits. Furthermore, in a comprehensive analysis, 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, since neither refers to market equilibrium. 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)*. Realisation of new energy saving measures would establish a new equilibrium, where the marginal costs equal the marginal willingness to pay, i.e. *p(2)* in figure 4. This is the only point where the two approaches yield the same result. The amount of energy saving is then *x(2)*.

In other words, both the damage-cost approach and the willingness-to-pay approach may give biased estimates. It is difficult to say which approach is the best, but in some cases one approach may be better than the other. If the supply curve is ‘flat’ compared with the demand curve, a marginal cost estimate would approximate the equilibrium price better than the willingness-to-pay estimate. The willingness-to-pay approach applies well with a ‘flat’ demand curve. A full analysis of the energy program requires a macroeconomic model, which includes a specification of the energy saving program, and relations between economic activities and environmental effects of energy saving. The remainder of this section is devoted to the bottom-up approaches. A macroeconomic model is developed in section 3.

2.3.2 The reduction in damage from air pollution.

Aunan et al. (1997) provide estimates of the reduction in damages caused by the emission reductions following an implementation of the Energy Program. The damages divide into impacts on health, on materials and on crops.

The three first columns of table 2 summarise the estimated impacts on health from implementing the whole energy program. The figures are based on effects in urban Hungary, which counts approximately 6.5 million people, or nearly 65 percent of the population.

The dose-response relationships are in general highly uncertain. The response estimates shown in the three first columns of table 2 are mainly based on studies of pollution changes in other countries. Many factors may cause variability of responses between countries. This adds to the uncertainty of the dose-response relationships. For instance, Aunan et al. (1997) estimated a 95 percent confidence interval to lie between 6 and 126 for infant deaths, and between 360 and 76 550 number of cases for chronic bronchitis. Hence, to base the use of environmental policy measures on responses indeed implies decision making under uncertainty. Below, however, we focus mainly on the expected responses.

Table 2 Estimated expected responses and assumptions of working days benefits per year from implementation of the energy program.

Symptom	**Expected response[1)]**			**Total increase in labour (man-years)**	
	Unit	Children	Adults	Patients	Health sector
1 Deaths[2)]	no.	34	70	74	11
2 Lung cancer	no.	-	25	50	4
3 Acute respiratory symptom	Days/person	1	0.15	1928	555
4 Chronic bronchitis	no.	14 040	16 520	3100	248
5 Asthma	Days/person	-	2.4	433	48

1) Source: Aunan et al. (1997)
2) For children, the number applies for infant deaths. For adults, under 65 years of age.

The economic benefits of the reduction of health symptoms following the energy program divide into the reduction in costs imposed on the health sector (hospitals doctors etc.) by increasing its relative capacity, and the reduction in costs related to those who are affected by pollution. The costs in the health sector could be based on estimates of the total cost per case in the health sector. The costs related to affected people accrue because they get sick and cannot work. In the cases were children get affected, they need someone to take care of them during their illness. Both can be expressed by the consequential loss of productivity.

Jacobsson and Lindgren (1996) have estimated the direct and indirect costs of all illness in Sweden, and provide estimates over total costs for the country as a whole. The direct and indirect costs of illness depend on several variables, which are difficult to assess within the frames of this study. For instance, the costs of health care include the costs of hospitalisation, consulting doctors in open health care and medicines. The distribution of these costs varies greatly among different symptoms, and among cases for a given symptom. Jacobsson and Lindgren (1996) found that hospitalisation accounts for 90 percent of the direct costs of cancer,[1] while they constitute less than 45 percent for respiratory diseases.

Direct costs also depend on how serious the illness is, but to our knowledge, no one has allocated direct costs according to the seriousness. The study of Jacobsson and Lindgren (1996) provides, however, a basis for such an allocation. This was done by adding some general assumptions about how the cost of death, disability and incidental illness divides into costs to hospitalisation, open care and medicines in each case. It was found that the average cost of death in the health sector is approximately 200000 USD for cancer and 1000000 USD for respiratory diseases in Sweden. Corresponding figures for disability was 10000 and 13000 USD and 3000 and 250 USD for incidental illness.

To transfer the estimates from Sweden to Hungary, the costs in the health sector were related to the sector's need for labour. The costs include, in addition, costs of medicines, hospitals, equipment etc. These are calculated as an overhead to the wages, and are assumed to amount to 1.5 times the wage bill.

Different principles may apply for an assessment of the costs of the loss of labour. A particular, and controversial, problem relates to the cost of death. Jacobsson and Lindgren (1996) base their estimate on the present value of the expected future earnings. This implies that the loss related to future potential earnings from one individual is included, but the "benefit" of having fewer mouths to feed is not. In a macroeconomic context, this means that the welfare increases as the population increases. This is not necessarily true.

In this study, we have based the estimate of the labour-cost on annual earnings. This leads to the problematic result that it is less beneficial to survive, than to avoid a disease that lasts for more than a year, such as cancer. We defend this choice by the aim of limiting the cost-estimates strictly to economic consequences, while considerations of 'reasonability' are taken care of by the willingness-to-pay approach.

The two last columns of table 2 show how the estimated responses from the Energy Program are assumed to affect the supply of labour and the capacity in the health sector in terms of labour. It is assumed that acute respiratory symptoms affect productivity by half a day per affected adult, and ¼ day per child per day of the symptom. People with chronic bronchitis are expected to have their working capacity reduced by 2 months per year, while people with children having this symptom have their working capacity reduced by 10 days per year. The health sector spends 1/10 days (3/4 of an hour) on acute respiratory symptoms per symptom day. Each case of chronic bronchitis is assumed to require 3.5 days per year from the health sector for children and 2.5 days for adults.

It is admittedly impossible to deal with 'the cost of deaths' as an indication of 'the value of a life'. It is difficult even when focusing only on the amount of money to spend on reducing the probability of dying from air pollution by one unit (the value of a statistical life). It is included here because deaths also affect the national product, although this effect is minor to many other aspects of death. The death of one adult is assumed to reduce the productivity per year

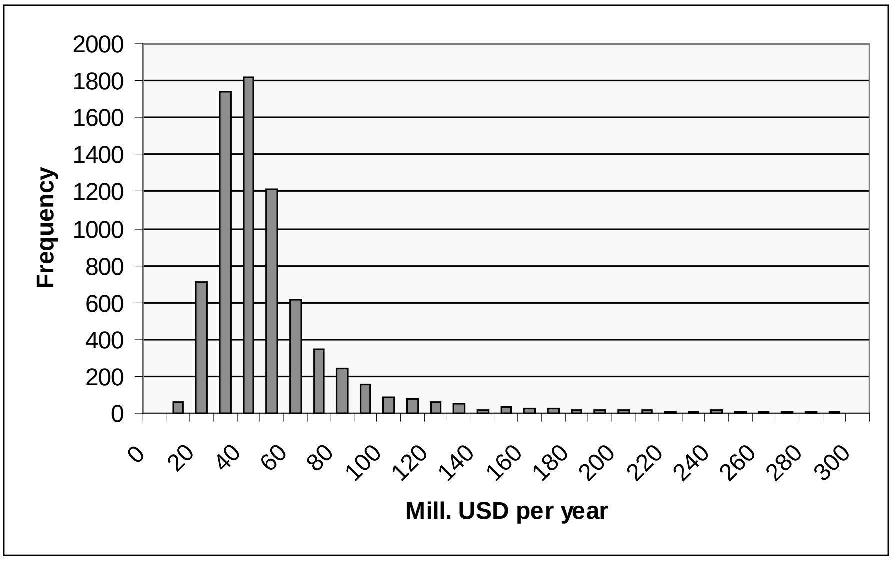
[1] This number applies for all kinds of cancer. In the calculations below, we consider lung cancer, only.

by one year. In the case of children, it is assumed that the parents' productivity is reduced by 45 days. In the hospital sector, the death of an adult requires 35 working days while the death of children requires 50. For cancer, each case affects labour productivity by 730 days per case, and takes 55 working days of the hospital's capacity. Hence, we assume that lung cancer means nearly 2 years off from work. Asthma is assumed to take ¼ of a working day per case, and 1/36 in hospital per case.

The total expected health benefits of the energy program were estimated to approximately 40.7 mill. USD per year. Note, however, that this estimate is quite uncertain. Figure 5 shows the results from a Monte-Carlo simulation of the health benefits based on the intervals provided for single effects, reported by Aunan et al (1997). A 90 percent confidence interval for these benefits ranges from 15.5 to approximately 130 mill. USD per year.

In addition to the health effects, a reduction of local pollutants limits the deterioration of materials, for instance by reducing corrosion, and affects the crops in agriculture. Aunan et al. (1997) estimated the benefits of the reduction in material damages to approximately 32 mill. USD per year, however, limited to the district of Budapest. The total damage on crops from pollution is considered to be substantial in Hungary, but the effect from the energy program on this damage is probably small - about 1 mill USD per year. The uncertainty of these estimates is not known.

Figure 5. Histogram of health benefits from energy program. 7500 draws.



The estimated expected total benefits of the energy program thus exceed the costs of the program by nearly 9 mill. USD if the alternative value of energy is set to zero. The health benefits, for which the uncertainty is indicated, could be lower than the cost of the program. Also the benefits on materials and the costs of the program are uncertain. Nevertheless, the benefits seem to be so substantial that it must be considered unlikely that the program will be socially unprofitable. Again, we emphasise that these estimates are based only on observable cost savings. As pointed out above, these savings may be considerably lower than what people are willing to pay for the environmental improvements. In particular, the damage costs of asthma and chronic bronchitis are probably low compared with the willingness to pay for reducing these symptoms. Hence, most studies such as Krupnick et al. (1996) and Aunan et al.

(1997) apply a combination of the two methods, dependent on what is considered to be the most representative for the value of the effects.

2.3.3 The willingness to pay for a reduction in damages.

In recent years, the willingness-to-pay-approach, or contingent valuation, has achieved much attention as a practical tool for decision making in environmental policy. The appropriateness of the approach has been intensively debated, see e.g. Arrow *et al.* (1993), McFadden (1994) and Hahnemann (1991). One of the main problems is that the approach is based on the perceptions of individuals, while market values can be revealed from actual behaviour. Another problem occurs when trying to generalise results of willingness to pay surveys by transferring estimates from one place to another, or to compare estimates between groups of people. To our knowledge, there are no applicable surveys of the willingness to pay for an improvement of the air quality in Hungary. To indicate the value of improved air quality by the willingness to pay approach, we therefore have to use estimates from other countries. Then, the problem of transferring estimates has to be dealt with.

As observed by Krupnick and Cropper (1992), this problem relates to what the point of reference is. From figure 4 it is clear that a statement from a person about the willingness to pay must refer to the initial situation $x = 0$. Little can be said about the willingness to pay if the environmental quality is different from $x = 0$, unless some assumptions are added. Then, what are reasonable and relevant assumptions? Krupnick et al. (1996) apply the results from surveys in industrialised countries, mainly the US, to a number of European economies in transition to assess the benefits of environmental improvements. They adjust the willingness to pay in accordance with income, on the background of assumptions about the income elasticity for ‘the environment’. For mortality, they also show the impact on the total benefit of applying an income-elasticity for mortality risk. What the income elasticity of environmental qualities is has been widely debated among economists. Intuitively, one should expect, such as McFadden and Leonard (1993), that ‘the environment’ is a luxury good, with income elasticity higher than 1. Kriström and Riera (1997), on the other hand, refer to several studies which indicate that the income elasticity of the willingness to pay is lower than 1, perhaps as low as 0.2.

As Flores and Carson (1997) point out, the income elasticity of the willingness to pay is different from the traditional income elasticity. To justify a transfer of willingness to pay estimates, and to apply them in a macroeconomic model, it is useful to relate the estimates to ordinary demand functions. Assume that avoided symptoms due to environmental policy are considered as a measure for the effect of improving the overall health status. We express this measure as an aggregate over the number of avoided cases for each of the five symptoms reported in table 2,

$$U(z) = \prod_{i=1}^{5} z_i^{\gamma_i}.$$

The quantity of each symptom, z_i, is measured for instance by ‘number of avoided cases’, and γ_i denotes how important each symptom is for an evaluation of the health status. We assume that γ_i across all symptoms adds to 1.The aggregate can be interpreted as a Cobb-Douglas utility function. By expressing health as an aggregate of different symptoms, it is assumed that the number of avoided cases of one symptom can be lowered at the expense of other symptoms without changing the overall health status.

A sound management of the total health expenditures implies that the efforts on each symptom are allocated such as to maximise $U(z)$. Denote by r_i the price, or the willingness to pay, for avoiding symptom i, and define the total expenditure on health as $G = \Sigma_i r_i z_i$. From the demand function for impact z_i we can express the willingness to pay for impact i as (see e.g. Varian, 1984):

$$r_i = \frac{\gamma_i}{z_i} G.$$

A transfer of adjusted estimates for r_i between countries is possible only if each symptom plays the same role in the aggregate, that is if γ_i is the same for both countries. Differences between r_i in two countries can therefore be explained by different health status, and by different expenditures. It is seen from the expression above that r_i can be adjusted proportionately to income only if the number of avoided cases per capita is equal in the two countries, and if the expenditure on health, G, is proportional to income (income elasticity equal to one).

Table 3 shows estimates of the willingness to pay in Hungary based on studies in the US under different assumption about these variables. Aunan et al (1997) assumed that the willingness to pay is adjusted in accordance with income. Hence, since the income in Hungary is reported to be 16 percent of a US income, the benefits are 16 percent of the US benefits for the same improvement of air quality. As noted above, this assumes that health status in Hungary is the same as in the US.

Table 3 The willingness to pay for the health effects of the Energy Program under alternative assumptions about the health status in the US and Hungary

Symptoms	**WTP per unit[1] (USD)**	**Estimated value of benefits from Energy Program (Mill. USD per year)**			
Income level		US	Hungary (30 percent of US)		
No of effects of pollution in Hungary relative to US		Same	Same (Aunan et al.)	Twice as many	Four times as many
Deaths[3]	4500000	468	76	112	131
Lung cancer	3000000	75	13	18	21
Acute resp. symptom[4]	98	198	32	48	56
Chronic bronchitis	240000	7334	1178	1759	2052
Asthma	36	22	3	6	6
Total	-	8098	1301	1943	2266

1) See table 2 for reference to unit
2) EPA (1995)
3) Deaths of people above 65 are excluded
4) Adjusted to take account for different degrees of seriousness

The two last columns of table 3 give the benefits under the assumption that the negative health effects of the air quality in Hungary is twice as bad or four times as bad as the effects in the US, respectively.[2] This affects the estimated benefits of the energy saving package considerably, and is nearly twice as high under the latter assumption.

[2] This requires that a reference level for pollution is given. We have assumed that the effects of pollution would be twice as high as today if no efforts to improve air quality were implemented presently in Hungary.

If we compare the cost-approach with the willingness to pay approach, we end up with benefits about 50 to 100 times higher when using willingness to pay for improved air quality to assess benefits. We believe these differences illustrate the methodological problems of the bottom-up approaches, but they may be related to other problems as well. We may have been too restrictive when making assumptions about the costs of health care, and, probably more important, the damage costs constitute only a part of the benefits from improved health quality. An indication of this is that the differences are largest for chronic illnesses and death. Finally, an important conclusion is that neither the cost approach nor the willingness to pay approach seems to be appropriate for an assessment of the benefits of the Energy Program.

2.4 The net benefit of the Energy Program

If the alternative price of energy is higher than 0.01 USD/kWh, the net benefit of energy conservation in the total Energy Program is positive, even if the value of less pollution is disregarded. If it is required that all the measures within the program should be beneficial, The required price is 2.7 cent/kWh. An alternative price of 0.5 cents requires that environmental benefits amount to approximately 17 mill. USD per year. This is well below the estimated benefits of reduced health costs and of corrosion costs.

Taking the benefits of less pollution into account, the calculations above indicate that the Energy Program should be launched irrespectively of the alternative price of energy, and of the method used to estimate the environmental benefits. When limiting the benefits to the damage costs of the pollution of energy use, the program turns out to generate a net social benefit. The annual cost of the program is 66.4 mill USD. A damage cost approach to the assessment of benefits indicates benefits of a better health status at 40.7 mill. USD, the benefits of less corrosion to 32 mill USD only in Budapest, and increase in crops to approximately 1 mill. USD per year. Regarded as a marginal project, the Energy Program is therefore beneficial, even if the alternative price of energy is zero.

The marginal benefits calculated by the damage cost approach is 1.16 mill. USD/PJ. The marginal cost of the program, defined as the unit cost of the most expensive measure, is 7.4 mill. USD/PJ. According to the damage cost approach, therefore, only a part of the program should be launched if the alternative price of energy is zero. Three measures exhibit unit costs lower than 1.16 mill. USD/PJ. These three measures conserve 48.9 PJ of energy.

The willingness to pay approach yields a total benefit of implementing the program at more than 1 900 mill. USD. This gives a marginal benefit of 30.4 mill USD/PJ, that is, well above the marginal cost of the Energy Program. When the difference between the damage cost approach and the willingness to pay approach turn out this large, it is hard to say which of the approaches give the best approximation to the actual value of less pollution. It may be that none of them gives a good estimate. Although both turn out with a net social benefit, the difference also contributes to increase the uncertainties about the Program. This is important in the context of decision making, since there are vast uncertainties about the costs of the program. Hence, there is a need to examine how market behaviour may change as a result of an implementation of the Energy Program and thereby contribute to a more accurate assessment of the value of environmental improvement.

3 A TOP-DOWN APPROACH

The basis for the top-down approach is a macroeconomic model. In this section, we demonstrate how the results from micro-studies can be applied in macro-models, and how macroeconomic relationships may change the conclusions given in the bottom-up analysis. At present, the Hungarian economy is in transition towards a market economy, and is still subject to significant market imperfections. Energy subsidies, particularly in the household sector, are relatively large, and there is a considerable informal sector in the country. Seip et al. (1995) suggest that this sector amounts to 30 percent of GDP, which is considered to be conservative. To carry out a realistic analysis of the macroeconomic impacts of the energy saving program requires in principle a model where the market imperfections in Hungary are modelled explicitly. However, this is beyond the scope of this paper. In order to demonstrate the linkages between micro-studies and the macroeconomic frames, it suffices to base the study on traditional assumptions about market behaviour.

This section starts with a presentation of the macroeconomic model. Then, the analysis of the impacts of the energy saving program is presented, and we conclude by comparing the results of this study with the bottom-up analysis.

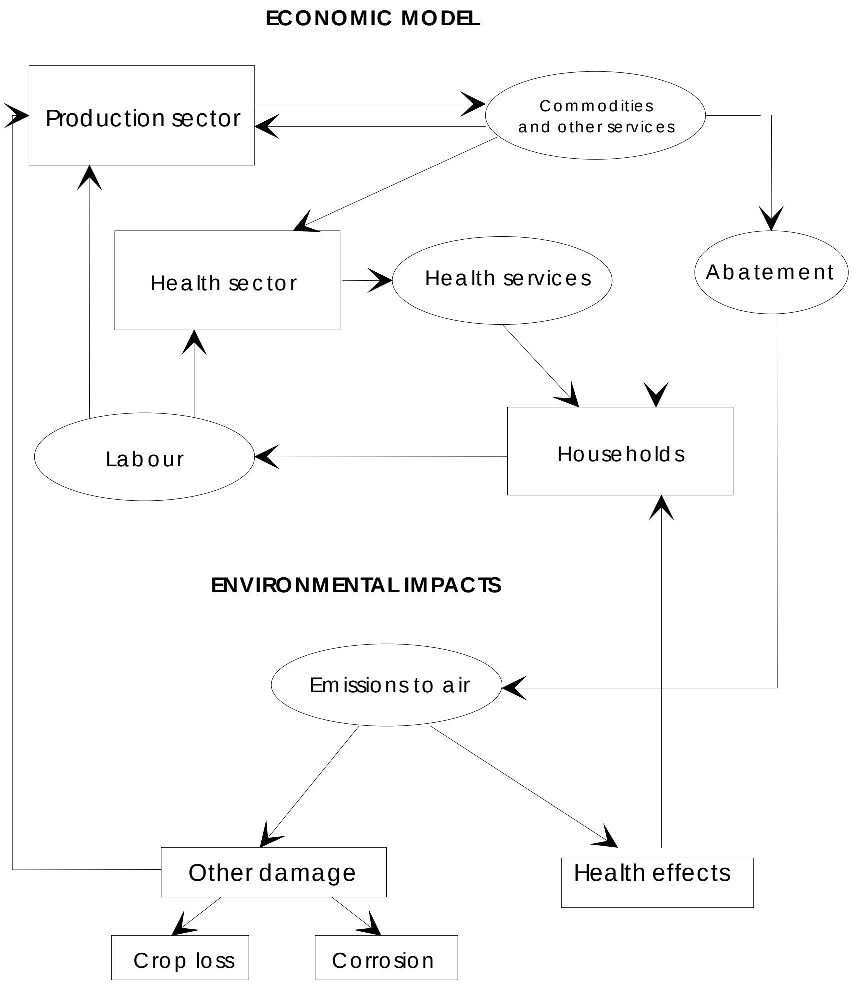
3.1 The Model

The calculations in section 2 are based on many assumptions about the impact on the environment from economic activities and how environmental change affects the economy. The macroeconomic model aims at making these relationships explicit and puts them into a common framework in order to study indirect effects. An analytic presentation of the model is given in the annex.

Figure 6 sketches the model. It consists of two interdependent modules, one economic and one environmental. The economic module includes three ‘sectors’, a production sector, a health sector and households. These are shown as boxes in figure 6. There are three activities in the economic module (ovals in figure 6), production of health services, production of an aggregate of commodities and other services, and use of labour. The arrows in the economic module indicate deliveries of commodities and services. Both the health sector and the production sector use labour and commodities and services as input. The households demand commodities and services (consumption) and health services. The model allocates the output of commodities and services, labour and health on the three sectors by general equilibrium conditions.

The model portrays the environmental effects of energy saving in a highly simplified manner. First, air quality is considered proportional to the instantaneous emissions from the production of commodities and services. The level of emissions is determined by two factors. One is the output level of commodities and services. The other factor is the abatement measures, which are represented by the Energy Program. Hence, emissions affect health and productive resources by corrosion and loss of crops.

Figure 6 Structure of the macroeconomic model



The 'demand' for emission reduction emerges from the explicit inclusion of dose-response-relations, indicated by long-dashed lines in figure 6. Productive loss is described by shifts in the production function resulting from changes in the emission level. These shifts can be interpreted as changes in the value of stocks, since corrosion affects the value of real capital and loss of crops is a result of deteriorated natural resources.

The impact of a changing health status is twofold. First, health affects utility directly, which means that the households demand health services to improve their wellbeing. Second, a better health status improves the supply of labour by increasing the number of working hours, thereby increasing the availability of productive resources.

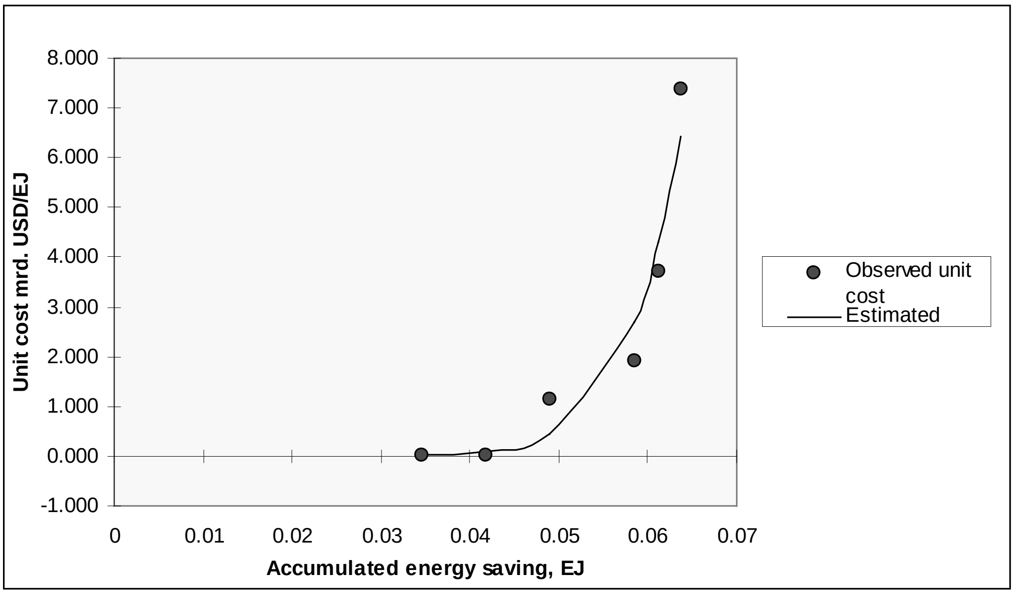
3.1.1 Application of the results from the bottom-up approach

It was estimated in the previous section that the Energy Program would enhance GDP by 32 mill. USD due to lower rate of corrosion in Budapest. We have limited the effect of corrosion to this amount and added 1 mill. USD in loss of crops. This gives 33 mill. USD as direct benefit in productivity due to less pollution. The model relates direct economic damage from emissions to the production in the commodity and service sector by adjusting the scale of production in the production function.

The health impacts of the Energy Program is represented in the model by an aggregate of the five symptoms studied in section 2. The ‘dose-response’-relationship thus gives the total number of reduced loss of man-years by patients, due to all the five symptoms, as a result of the energy conservation. For simplicity, it is assumed that the number of reduced loss of man-years is proportional to the amount of energy saving. The number of lost man-years from the working force is then subtracted from the total stock of labour.

The loss of man-years spurs activity in the health sector. The costs of supplying health services in terms of labour costs and ‘overhead’ are represented by the demand for labour and commodities and services, respectively. The level of activity in the health sector is determined by the equilibrium conditions. This means that the health sector adjusts its capacity to changes in the demand for health services immediately.

Figure 7 Marginal abatement cost curve based on the Energy Program



The possible increase in the supply of labour constitutes one part of the value of a better health status. The other part relates to the direct welfare effect of an improved health status. We have assumed that this effect is related to the risk of getting affected by some disease. This risk is not explicit in the model, but is explained by the emission reductions. Instead of demanding a lower risk for getting affected, we therefore assume that the households demand emission reductions. The value of these reductions is taken from the estimated total willingness to pay in the bottom-up calculations. Taking the figures in table 3 literally, a budget relation for the households may be constructed and implemented in the model. If we assume that the health status related to air pollution in Hungary is twice as bad as in the US, table 3 shows that Hungarian households are willing to spend 1.9 billion USD per year to implement the Energy Program.[3] The initial unit value, or price, of each PJ conserved can then be defined implicitly.

[3] The total budget for households in Hungary is approximately 30 bill. USD. Consequently, our calculations indicate that the households would be willing to spend more than 6 percent of its income on the Energy Program.

Finally, the Energy Program is represented by a separate abatement cost function in the model. This function is based on the ranking of the measures included in the Energy Program. Figure 7 displays the function and the 'observed' costs of the different measures. Recall, however, that the 'observations' are based on a number of assumptions since there is little information about the costs of the single measures. The cost curve is characterised by a large potential of energy saving at nearly no costs. These are due to the effect of energy awareness described in the program. Beyond 40-50 PJ, the cost curve becomes very steep.

Note that the cost curve relates only to the measures of the Energy Program. Other measures to reduce emissions of local pollution may also be available in Hungary. A representative curve for all measures may therefore differ from the one shown in figure 7. It is also important to notice that the cost curve in figure 7 may be prolonged beyond the measures that constitute the Energy Program, i.e. 63.7 PJ. Market equilibrium gives an amount of abatement at a cost corresponding to a point of the cost curve, regardless of whether measures in addition to those actually proposed in the Energy Program are needed or not. Below, 'the Energy Program' refers to the measures actually constituting the Hungarian proposal, while the 'Energy Program curve' denotes any point of the cost curve displayed in figure 7.

3.2 Evaluation of the Energy Program in a macroeconomic context

Most macroeconomic studies of the national costs of emission control have been focusing on CO_2-emissions. For moderate reductions in CO_2-emissions up to year 2020 the studies estimate the cost to be between 0.025 and 0.075 percent of the world's GDP per percent reduction in emissions (see e.g. Barns *et al*., 1992, Manne, 1992 and Oliveira Martins *et al*., 1992). For high emission reductions, 30 to 50 percent, the estimated cost per percent reduction in CO_2-emissions is somewhat higher. However, few studies include the benefits of the reductions in local and regional pollution associated with the reduction in CO_2-emissions.[4] This indicates that the costs of climate policy may be significantly overestimated.

The social cost of pollution control is usually calculated as the deviation from the initial equilibrium with no restrictions on the emissions. Clearly, an emission target implies a positive, social cost if no improvement in health and environment is represented in the model. This means that measures yielding net social benefits, so-called no-regret options, are not available. Since the aim of evaluating the Energy Program partly is to point at no-regret options, two states of the economy are compared in the present study. The base case represents the initial situation where the commodity and service market is in equilibrium, and the Energy Program is not implemented. This is the point at which the model is calibrated. There is no relation between the activity in the health market and the demand for emission reductions in the base case.

In the alternative case, the Energy Program, as described by the cost curve in figure 7, is implemented. Implementation of the Energy Program could be analysed in two ways. One is to require that the actual emission reductions are equal to the demand for emission reductions.

This seems to be unrealistic, and indicates that the health aggregate has a different functional form than assumed, and/or that Americans and Hungarians have different welfare functions. A sensitivity analysis of this assumption is presented below.

[4] Exceptions are Ayres and Walter (1991), Alfsen et al. (1992), Barker (1993), see also Aunan et al. (1997).

This alternative is denoted equilibrium in table 4. A second approach is to introduce a charge on the polluting commodity, and require that the marginal cost of abatement is equal to the charge rate (denoted charge in table 4). The first approach corresponds to the basic idea in this paper. We also present results from the second approach, because it is the most frequently used in top-down studies of the costs of CO_2 emission control.

Table 4 presents the main results. According to the calculations, an implementation of the Energy Program will enhance GDP by 301 mill. USD, or 0.3 percent according to the equilibrium condition. Although negligible in a macroeconomic context, it nevertheless indicates the social profitability of the Program, and must be considered large compared with the annual cost of the program, which amounts to 66.4 mill USD.

The reduction in households' consumption of ordinary commodities and services is compensated by a better health status, which amounts to more than 6 100 man-years. This increase in the supply of labour is channelled to the commodity and service sector, which leads to higher output. Hence, this sector needs to use more of its own output as input. The remaining part of the reduction in households' consumption is spent on the measures in the Energy Program.

Table 4 Comparison of main indicators between base case and alternatives

	Unit	Equilibrium	Charge
Increase, GDP	*Mill. USD*	301	394
Increase in use of C&S in households[1]	*Mill. USD*	- 775	- 708
Increase in labour supply	*Man years*	6 138	5 181
Total abatement costs	*Mill. USD*	187.0	37.7
Energy conservation	*PJ*	73.7	63.7
Emission reductions	*Percent*	7.4	6.1

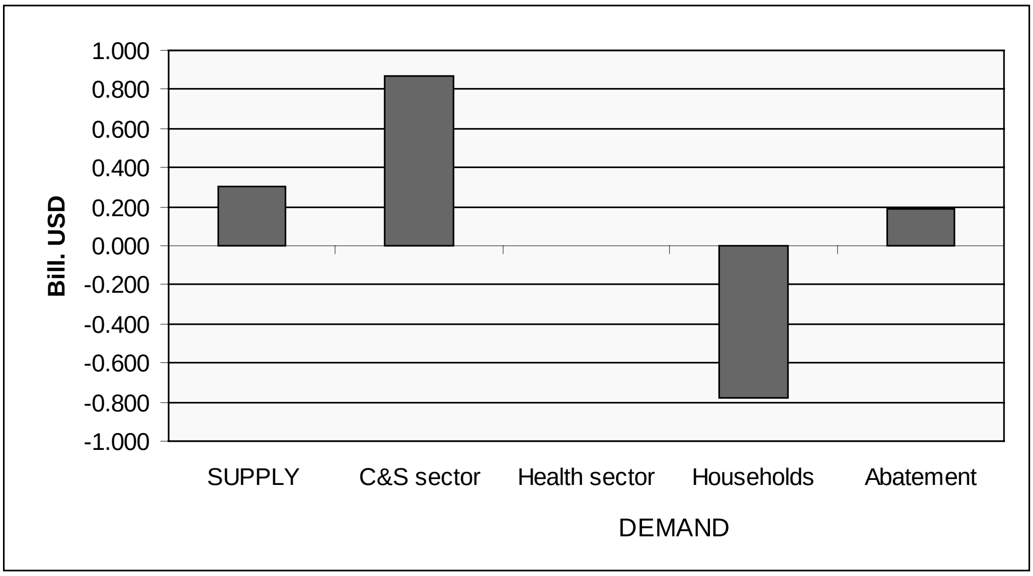
1) C&S is 'consumption and services'

The last column of table 4 shows the impacts of a charge exactly sufficient to ensure implementation of the Energy Program. The charge rate turned out to be 0.075, or 7.5 percent of the basis price of commodities and services. The effect of a charge is likely to be small. This is because pollution is related to the output of the commodity and service sector, which constitutes nearly 99 percent of the total GDP in the model calculations. The possibilities of reallocating resources are therefore extremely limited in the model. As a consequence, it is difficult to obtain quantitative targets, such as emission targets, by means of a shift in relative prices initiated by a charge, for instance. The charge has limited impacts on the macro economy in the model. It may seem surprising that the increase in GDP is higher in the case of a charge than in equilibrium, which represents an 'optimum'. This illustrates, however, one of the problems of using GDP as a welfare measure. The increase in GDP in the charge case follows from the fact that the demand for emission reductions exceeds actual emission reductions. As a consequence, the abatement costs are lower, and this leaves more resources to be allocated among the three sectors, households, health sector and production of commodities and services. The increase in the two latter sectors enhances GDP.

The total abatement cost in the charge alternative is only about half of the cost of the Energy Program, (37.7 vs. 66.4 mill. USD per year). This is because the cost function does not fit exactly to the observations of the costs of measures, see figure 7. Thus, while the total cost of the Energy Program was estimated to be 66.4 mill. USD per year, the total costs according to the Energy Program curve is 37.7 when energy conservation equals 63.7 PJ. When considering the social benefits of implementing the Energy Program in the macroeconomic model, we should therefore compare with an annual cost of 37.7 mill. USD.

Total abatement costs in equilibrium is calculated to be 187 mill. USD per year. This is more than twice the calculated reductions in environmental costs of the five grouped measures suggested in the Energy Program, but only 10 percent of the estimated willingness to pay for the bottom-up analysis carried out in chapter 2. The 187 mill. USD corresponds to a marginal cost of abatement of 27.4 mill USD/PJ (*p(2)* in figure 4, section 2.3.1). The marginal cost of the Energy Program, i.e. the unit cost for the most expensive measure, is approximately 7.4 mill. USD/PJ. The marginal cost of abatement (or the shadow price of emission reductions) at 27.5 million allows for 74 PJ to be conserved by direct abatement, if the cost-function actually represents the abatement cost curve. Hence *x(2)* in figure 4 is 74 PJ. This answers the dilemma illustrated by figure 4, whether the cost approach or the willingness to pay approach is the most relevant for the evaluation of the energy program. The large deviation between the two estimates of the benefits described in chapter 2 was partly explained by the different environmental aspects covered by the damage cost estimate and the willingness to pay estimate. The bottom-up approach can therefore be considered as an essential starting point, but to combine the different issues and obtain a realistic estimate, a macro-economic model may be necessary. This is particularly important in cases where the supply and demand curves are steep, and small shifts may lead to significant changes in relative prices.

Figure 8 Changes in supply and demand for goods and services by sector. Bill. USD



From an economic point of view, it is important to know how the resources are reallocated when equilibrium in the health market is established and the Energy Program is implemented. This is shown in figure 8. The input to the commodity and service sector and to abatement is increased, while consumption in households is reduced. This reduction is due to household's increase in the demand for health services.

Emission reductions resulting from specified measures such as those included in the Energy Program are often regarded as contributions to national targets. In the case of the Energy Program, this means that an investing country could pay the bill to Hungary, and thereby reduce own emissions by nearly 5 mill. tons of CO_2. The results reported in table 4 show, however, that the output of commodities and services increases. The emissions increase as a consequence. In other words, the national emission reductions deviate from the reductions given by the Energy Program curve taken in isolation. Regarded as a climate measure, the implementation of the Energy Program therefore "leaks" by approximately 5 percent, according to the calculations. It is, however, difficult to say to what extent this is due to the level of aggregation of the model. A closer study of the leakage demands an assessment of the substitution between sectors following an implementation of emission targets. This requires a less aggregated model.

As pointed out, the estimates in this study are based on a large number of assumptions. One of the main problems has been to attach cost estimates to the measures included in the Energy Program. Only the total investment for the entire program was available. This was allocated to the different measures on a rather arbitrary basis. Nevertheless, this allocation constitutes the basis for the cost curve applied in the model. A second problem is the uncertainty of the dose-response-relationships, which was discussed in relation to figure 5. Thirdly, it was pointed out that it is by no means straightforward to transform estimates of the willingness to pay in one country to another. This adds to the general difficulties in obtaining reliable estimates of the willingness to pay.

Table 5 Comparison of main economic and environmental indicators between base case and alternative cost curve and willingness to pay

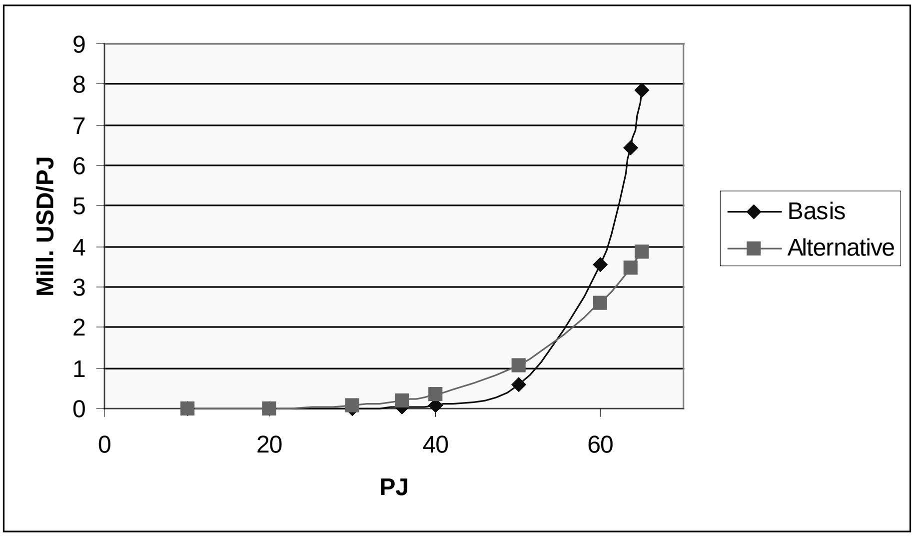
	Unit	**Equilibrium case**	**(i) Moderate abatement cost**	**(ii) Lower WTP**
Increase, GDP	*Mill. USD*	301	220	-321
Increase, consumption of C&S[1]	*Mill. USD*	- 775	-821	-187
Increase in labour supply	*Man years*	6 138	7 723	5 773
Total abatement costs	*Mill. USD*	187	1 922	29
Energy conservation	*PJ*	73.7	91.2	62.2
Emission reductions	*Pct*	7.4	9.5	6.9

In table 5, the willingness to pay and the abatement cost functions are changed to see how such changes affect the results. For the sake of comparison the basis equilibrium solution from table 4 is also included.

In the first alternative (i), the variety in the unit costs of the measures is assumed to be less than in the basis alternative (see figure 9) The least costly measures become more expensive, while the most costly measures cost less than in the basis alternative. For instance, the marginal abatement cost of 'awareness' is 0.02 million USD/PJ in the basis alternative and 0.2 million USD/PJ in alternative (i), while the respective marginal costs for 'insulation and renewables' are 7.9 USD/PJ and 3.5 USD/PJ. All measures beyond those explicitly included in the Energy Program are therefore expected to be less costly in alternative (i) than in the basis alternative. The cost curves in the two alternatives are compared in figure 9.

More moderately increasing marginal abatement costs lead to a reduction in the shadow price of energy conservation from 27.5 mill USD/PJ to 20.5 mill USD/PJ. Since the marginal cost curve is more gently sloping, the amount of energy conservation increases from 73 to 91 PJ. The total cost of abatement at optimum thereby increases considerably, from 187 million USD to 1.9 billion USD per year, that is, approximately the same as the willingness to pay for the Energy Program in the bottom-up analysis. This leads to a better health status and an increase in the labour force by more than 7 700 man years.

Figure 9 Marginal abatement cost curves for the Energy Program with alternative variability in the unit costs between measures.



The increase in total abatement leads, however, to a relative shortage of commodities and services, if compared with labour. At the same time the supply of labour increases. This is partly because less people are affected by pollution related diseases. As a result of this, the price of health services decreases, and the activity in the health sectors goes down. As a consequence, the wages remain approximately the same, and consumption is only slightly affected, compared with the equilibrium case. In the commodity and service sector, there is a tendency to substituting input of commodities and services for labour, but not sufficient to compensate for the increased demand for these goods into abatement. Hence, the output from this sector and GDP is reduced. In short, therefore, more moderately increasing marginal abatement costs lead to a better health status, less consumption, and some GDP than in the equilibrium case. Compared with the basis case without the Energy Program, however, GDP increases, which contributes to an increase in the energy consumption. The energy conservation on the national level is 88.5 PJ, which means that the "leakage" is about 2.7 PJ.

In the second alternative, (ii), the willingness to pay for a lower risk of health impacts is reduced to ¼ of the willingness to pay assumed in the basis equilibrium solution. This means that the willingness to pay for the improvement caused by the measures explicitly included in the energy program is reduced from nearly 2 billion USD to nearly 0.5 billion USD per year, constituting about 1.7 percent of their household budget. We assume that the rate of substitution between commodities and emission reductions remains the same.

This results in a radical drop in the shadow price of emission reductions, which reduces the efforts in the Energy Program to 62.2 PJ. This is approximately the same as the 63.7 PJ given for the Energy Program. The shadow price of energy conservation in this alternative is 5.1 mill USD/PJ, which is lower than the most expensive measure in the Energy Program. The relative shift in consumers' demand also implies an increase in consumption goods, which leaves less for input in the commodity sector, and a lower output from this sector. Also the activity in the health sector is slightly increased, due to higher emissions, although higher emissions compared to the "equilibrium case" are counteracted to some extent by a lower output of commodities and services.

The total cost of abatement measures in alternative (ii) at 29 mill. USD is less half of the estimated cost of the energy program (66.4 mill USD). This shows that it is not necessarily optimal to carry out the whole Energy Program, even if the net benefit calculated by the damage cost approach is positive. In reality there are additional opportunities to reduce emissions; in the present macroeconomic model represented by a reduction in output. The total reduction is therefore about the same as estimated for the whole energy program in the bottom-up approach.

4 CONCLUSIONS

Estimates of the benefits of less pollution may differ significantly dependent of the choice of method. Bottom-up approaches, using damage costs or willingness to pay assessments and top-down approaches are therefore often considered to be alternatives. The main conclusion is that the approaches support each other, and that all should be carried out in order to do an appropriate assessment of the benefits. The damage cost approach provides vital information about the economic consequences of reduced emissions to air. The willingness to pay approach is necessary to include the demand for improved environmental quality, and the top-down approach is needed to find the cost-efficient level of emission cuts.

The Energy Program discussed in this study is originally meant as a measure to reduce the emissions of CO_2. From an economic viewpoint, however, the social profitability of the program seems indisputable because of the reduction in energy costs following the energy conservation. Moreover, and not at least because of the health effects, the benefits of lower emissions of SO_2, NO_x and particles are probably larger than the benefits of reducing CO_2-emissions.

The value of the benefits from less pollution exceeds the reported cost of the Energy Program regardless of the approach used for estimating the benefits. The cost-figures are, however, highly uncertain. As expected, the willingness to pay approach gives higher benefit estimates than the damage cost approach. The willingness to pay approach gives between 20 and 40 times higher benefits than the damage cost approach. The large difference may partly be due to the fact that the willingness to pay estimates are transferred from other countries, notably the US. To provide more reliable estimates of the demand for lower emissions, a study should therefore be carried out in Hungary.

According the top-down study, Hungary could benefit from implementing more energy conservation than the amount suggested in the Energy Program. How much depends critically on the costs of measures not included in the Energy Program. If the marginal cost of energy conservation increases less than assumed in this study, another 30 PJ may be conserved at a net social benefit. On the other hand, a lower willingness to pay than assumed here may also be realistic. Nevertheless, the energy program is beneficial, even with a radical reduction in the willingness to pay for emission reductions.

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ANNEX Model for Comprehensive Integrated Analysis (CIA)

This is a macroeconomic equilbrium model for analysis of environmental benefits of specific measures in a comprehensive framework. The model has three sectors. One sector produces ordinary commodities and services, there is a health sector and households. The activity in the production sector affects the envionment by emissions which lead to environmental damage. The emissions can be reduced, either by reducing output or by direct abatement. The damages include material damage on the stock-resources utilized in the production sector (material damage and crop loss), effects on human health and a consequential cost imposed on the health sector.

Production sector

Gross output, x, of ordinary goods and services exhibits constant elasticitiy of substitution between input of ordinary goods (x_1) and labour (n_1) (CES-technology):

$$x = (b + \beta_1 x_1^{\rho_1} + \beta_2 n_1^{\rho_1})^{\frac{1}{\rho_1}}; \tag{1}$$

The constant term, b, can be interpreted as the contribution to gross output from stock-resources, such as capital and natural resources. The stock-resources are affected by pollution via material damage and damage on crops. Let B denote the stock value of these resources and denote the total reduction in emissions by dm. We assume a linear 'dose-response-relationship' for material damage, that is,

$$b = (1 - \beta_1 - \beta_2)[B(1 + \xi dm)]^{\rho_1} \tag{2}$$

Denote by w, wages and p, the buyer price of ordinary goods and services. The demand for ordinary goods and services in a CES-technology can be written (see e.g. Varian, 1984):

$$x_1 = (x^{\rho_1} - b)^{\frac{1}{\rho_1}}[(\frac{p^{\rho_1}}{\beta_1})^{\frac{1}{\rho_1 - 1}} + (\frac{w^{\rho_1}}{\beta_2})^{\frac{1}{\rho_1 - 1}}]^{-\frac{1}{\rho_1}}(\frac{p}{\beta_1})^{\frac{1}{\rho_1 - 1}}; \tag{3}$$

Correspondingly, the demand for labour is:

$$n_1 = (x^{\rho_1} - b)^{\frac{1}{\rho_1}} [(\frac{p^{\rho_1}}{\beta_1})^{\frac{1}{\rho_1 - 1}} + (\frac{w^{\rho_1}}{\beta_2})^{\frac{1}{\rho_1 - 1}}]^{-\frac{1}{\rho_1}} (\frac{w}{\beta_2})^{\frac{1}{\rho_1 - 1}}; \qquad (4)$$

Health sector

The health sector uses labour and ordinary goods and services to produce its output, z. We assume a Cobb-Douglas technology for providing the required demand. The supply of health-service z is:

$$z = a n_2^{\mu} x_2^{(1-\mu)} \qquad (5)$$

where x_2 is the use of goods and services and n_2 is labour. z is a production index, here measured as a proportion of the number of working days lost due to 'pollution'. The factor relates to the assumptions made on the capacity required in the health sector for each symptom. The demand for x_2 required to produce this service can be written (see e.g. Varian, 1984):

$$x_2 = \frac{z}{a}(\frac{w}{p})^{\mu}(\frac{1-\mu}{\mu})^{\mu} \qquad (6)$$

and the demand for labour is:

$$n_2 = \frac{z}{a}(\frac{p}{w})^{1-\mu}(\frac{1-\mu}{\mu})^{1-\mu} \qquad (7)$$

Households

The households are assumed to derive utility from an aggregate of ordinary commodities and services and from an indicator for the health status. This indicator may be expressed for instance as the probability of getting affected by environmentally caused diseases. To simplify, we assume that this probability is a linear transformation of emission-reductions. Hence, the utility can be expressed by a relation to the consumption of ordinary commodities and services and emission reductions. The utility is expressed by constant elasticity of substitution (CES-utility) between ordinary goods and services and emission reductions. The budget constraint is:

$$R = px_c + qy_c;$$

where R is the total income in households, p and x_c are price and quantity of ordinary goods and services, respectively, and q and y_c are the price and the quantity of the demanded emissions reductions, respectively. Then, the demand for ordinary goods and services can be written (see Varian, 1984):

$$x_c = (\frac{p}{\alpha})^{\frac{1}{\rho_2 - 1}} \frac{R}{\alpha^{\frac{1}{1-\rho_2}} p^{\frac{\rho_2}{\rho_2 - 1}} + (1-\alpha)^{\frac{1}{1-\rho_2}} q^{\frac{\rho_2}{\rho_2 - 1}}} \tag{8}$$

and the demand for emission reductions is:

$$y_c = (\frac{q}{1-\alpha})^{\frac{1}{\rho_2 - 1}} \frac{R}{\alpha^{\frac{1}{1-\rho_2}} p^{\frac{\rho_2}{\rho_2 - 1}} + (1-\alpha)^{\frac{1}{1-\rho_2}} q^{\frac{\rho_2}{\rho_2 - 1}}} \tag{9}$$

The demand for emission reductions from households cannot be observed directly, since these emission reductions are not bought and sold in markets. However, it was assumed that the demand for emission reductions is due to the health impacts. Thus, the total willingness to pay for health improvements is the same as the willingness to pay for emission reductions. To set the parameters of the demand functions, we therefore utilise the reported estimates for the willingness-to-pay for reducing the frequency of the different symptoms. Assume that improvements in the health status can be represented by an aggregate of the five symptoms, z_j:

$$U(z) = A \prod_{j=1}^{5} z_j^{\gamma_j}$$

This aggregate is based on an assumption that the households can substitute between the different symptoms. Hence, if the price of one service increases relative to the others, the households will demand less of this service and more of the other. Denote by r_j the willingness to pay for the reduction in one case of symptom j. The total expenditure on health services is:

$$G = \sum_{j=1}^{5} r_j z_j \tag{10}$$

By minimisation of G for a given level of $U(z)$, we find the demand for health care of symptom j:

$$z_j = G\frac{\gamma_j}{r_j} \tag{11}$$

Note that estimates of the willingness to pay for environmental improvements can be decomposed on the basis of this expression, since it follows that $r_j = \gamma_j G/z_j$. Hence, if we believe that the utility function is indentic in different countries, the parameters, γ_j, can be estimated by use of surveys over the willingness to pay for health improvements in other countries. As we can see, this requires knowledge of both G and z_j in Hungary as well as in the country from which the estimates are taken. In this study, G and z_j are based on rough assumptions. To the macroeconomic model, we need to use only $G = \sum r_i z_i$. By the use of G we can implicitly assess the parameters of the demand function.

The households' income is defined by the total wage-bill of the economy:

$$R = wn \tag{12}$$

The supply of labour is restricted by the total available work force, $\bar{n}$, and the extent to which people are affected by pollution. In this model we limit the focus on measures to a 'one-dimensional' energy saving package, and do not raise the question of how to rank measures according to effects. This allows for a very simple representation of 'dose-response-relations' for health impacts. For simplicity, we express the responses in terms of loss of working days. This is clearly a simplification, but the detailed description of health effects esimated for the Energy Program makes it nevertheless acceptable. The 'dose-response-relationship' expresses the relation between loss of working days and emissions

$$z = Sm^{\sigma} \tag{13}$$

where z is the number of working days lost due to pollution. In equilibrium, this equals the output of the health sector. Total supply of labour is

$$n = \bar{n} - z \tag{14}$$

Market equilibrium and emission control

There is no observed price of services produced in the health sector since the demand for emission reductions is only indirectly related to the supply of health services. Consequently, the budget constraint in households does not follow from all the equilibrium conditions. This means that all the equilibrium conditions must be made explicit, and that the absolute prices are determined.

The objective of environmental policy is to control the emissions to air. We assume that emissions are linearly dependent on the output in the production sector (alternatively, the total input in all sectors), but can be reduced by implementing energy saving measures, x_a:

$$m = \epsilon x - G x_a^{\eta} \tag{15}$$

where the last term is the inverse cost-function for abatement measures.

Emission reductions demanded by households are measured relative to the known, inital level $\hat{m}$. Hence,

$$y_c = \hat{m} - m \tag{16}$$

The model can be closed in two alterantive ways. The basic idea is to close it by means of equilibrium between the demand for emission reductions and emission reductions resulting from implementation of energy conservation measures. Then, the marginal cost of abatement is to be equal the price of emission reductions:

$$q = G x_a^{\eta}$$

The alternative, which is the standard way in macroeconomic analyses of environmental control, is to impose a charge, t, on the polluting activity. In order to achieve a cost effective environmental policy the level of t and x_a should be chosen such that the marginal cost of abatement equals the marginal cost of charging emissions. The charge for one unit of emissions is t/ε. The condition for cost effectiveness is therefore

$$t = \varepsilon G x_a^{\eta}$$

The charge constitutes the difference between the seller price and the buyer price of the commodity and service aggregate, i.e.

$$p_{net} = p + t \tag{17}$$

where p^{net} is the seller price of the product and service aggregate.

There are three markets, the health market, the 'ordinary' product and service-market and the labour market. Equilibrium in the product market requires that

$$x = x_1 + x_2 + x_c + \int_0^{x_a} G\chi^{\eta} d\chi \tag{18}$$

Equilibrium in the labour market implies

$$n = n_1 + n_2 \tag{19}$$

(5) and (13) define z by supply and demand, thus assuring that the health market is in equilibrium. Note that the shadow price of health services is the numeraire.

This is CICERO

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

The research concentrates on:

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

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

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

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