The value of the environment – Is it a matter of approach?

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This paper discusses why estimates of the benefits of reduced air pollution differ in accordance with the approach used. Estimates based on bottom-up studies of the damage costs related to air pollution usually turn out much lower than estimates based on assessments of the utility of reduced air pollution, obtained for instance by willingness to pay assessments. This is usually explained by the fact that the willingness to pay approach includes the utility aspect of non-market values, and for this reason, it is often preferred to the damage cost approach. This is, however, not the whole story. The paper shows why alternative approaches should not be considered as being in conflict, but rather as means to get supplementary information necessary to put a value on environmental quality. Information from bottom-up assessments of damage costs and from studies of the willingness to pay is used in a macroeconomic model to carry out an evaluation of the social costs of energy saving measures in Hungary.

Keywords: environmental economics, bottom-up/top-down approaches, secondary benefits

1. Introduction

That the benefits of mitigating environmental deterioration may exceed the costs of the abatement measures has been recognised at least since 1875, when Smith [25] claimed that it would be economically beneficial to reduce the emissions along the river Tyne. Concern for the increased need for cloth washing in British households due to the dirt caused by emissions from the industry led Pigou [22] to write his seminal paper on optimal taxation. He showed that a charge on the emissions corresponding to the marginal loss of utility for the households could be interpreted as the social cost of the emissions. Largely due to Pigou's early contribution, Baumol and Oates [7] could commence their textbook on environmental economics with the following claim: "When the "environmental revolution" arrived in the 1960s, economists were ready and waiting".

Although charges play an increasingly important role, environmental policy making goes beyond the assessment of charges. Policy makers continuously 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. By use of different methodologies, one may end up with widely different estimates of benefits for the same improvement.

This is largely due to the different points of departures taken in the alternative methods. In some cases, the benefits are estimated with reference to the costs of damage caused by environmental deterioration. There are different ways to make such assessments (see, e.g., [26]), for instance by estimating the cost of damages (e.g., [4]), or the costs of avoiding the damage. Another approach is to assess the demand for environmental improvements. Again, a number of methods have been suggested. One approach is to ask people what they are willing to pay [18,19], another to develop utility functions that include environmental aspects by the use of expert panels (e.g., [27]). Yet another method is to reveal the demand from observed market behaviour by hedonic pricing [8,23], or by implicit valuation methods [10].

Krupnic et al. [15] suggest that assessments of damage costs and demand for improved environment differ with a ratio of 1 : 2–1 : 3. This is often explained by the fact that demand-side assessments include appreciation of non-market values. Strictly speaking, however, there is no consistent link between the two approaches, and the ratio may greatly exceed the given limits. One reason may be that if environmental policy is sub-optimal, the willingness to pay does not reflect the marginal cost of improving the environment. The second reason may be that the valuation is carried out prior to the implementation of the measures and is based on constant prices, such as in bottom-up studies. However, the prices may change as a result of the implementation of the measures. There is no general answer as to whether estimates based on marginal costs are better or worse than estimates based on the 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. An appropriate assessment of environmental values requires that both the damage costs and all the relevant factors determining demand are examined. In addition, this information should be implemented in a macroeconomic framework in order to find the equilibrium price. In this paper, we develop a model for integration of damage costs and demand side assessments in order to estimate the value of environmental improvements. The focus is on the methods, and the results are based on a highly simplified economic model, inappropriate for practical decision making, but sufficient to demonstrate the impact of alternative choices of methodology. The example is taken from a proposed program for energy saving in Hungary, henceforth called the Energy Program. The program was presented by the Hungarian government in their communication to the Framework Convention of Climate Change [21]. Thus, it was suggested as an initiative to reduce emissions of CO_2, but the major effects relate to the reduction of local pollutants (see [2,6]). In this paper, we study the economic aspects of the Energy Program, and consider the physical effects on air quality and impacts on health, material damage and crops as given. A detailed discussion of the assessments of the physical effects on health, materials and crops from the Energy Program can be found in [6].

2. Approaches to the valuation of environmental benefits

Figure 1 illustrates the difference between the alternative approaches of valuation used in this study. The thick MC-curve represents the marginal cost of the measures, for instance those included in the Energy Program. In this case, the costs are measured in terms of cost per saved PJ of energy. Assume that we aim at evaluating a set of measures that leads to a reduction in energy consumption equal to $x(1)$. The marginal cost of this program is $p(1)$. In practice, this could represent the unit cost of the most expensive measure within the set of measures. Most often, cost benefit analyses evaluate a whole set of measures within a given investment proposal. Then, the costs are equal to the dark shaded area below the MC-curve, and the cost-benefit criterion is whether or not the whole set of measures yields higher benefits than costs. This is strictly correct only if the unit costs are independent of the size of each measure, and the same for all measures. If this is not true, the test is how much one could invest in each measure before marginal costs exceed marginal benefits.

The damage-cost approach is based on the idea that the environmental benefits associated with reduced emissions should be subtracted from the marginal costs. For example, reduced air pollution leads to an improvement of the health status, thereby increasing the productivity of the population and reducing the costs of health care. The marginal cost curve for the abatement thereby shifts downwards, e.g., from MC to the marginal social cost curve, MSC in figure 1. Energy saving might therefore 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 the figure. In the example displayed in figure 1, inclusion of environmental benefits turns the marginal social cost of the program negative, a so-called no-regret option.

Assume now 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 a total benefit equal to the total of the light and the dark shaded areas in figure 1. This is not a perfect criterion, since the willingness to pay and the marginal cost

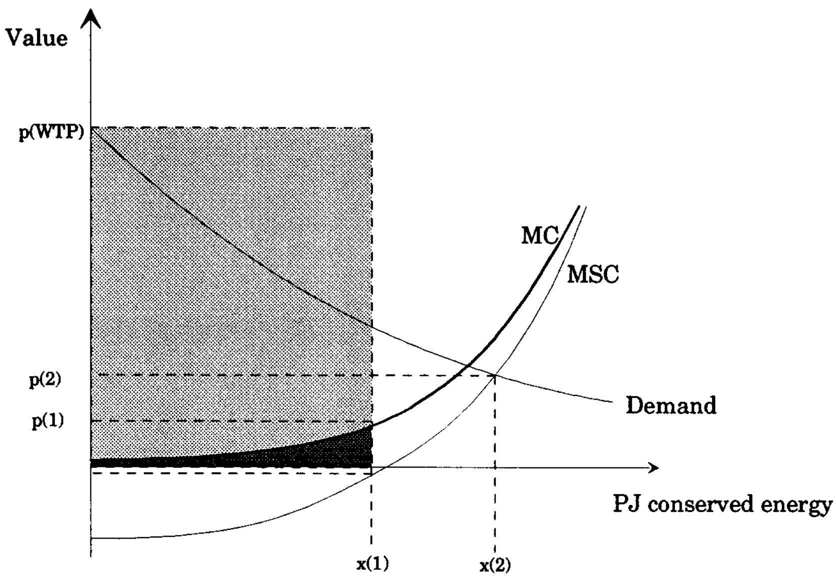

Figure 1. Approaches to environmental valuation.

refer to different quantities of energy conservation, $x = 0$ and $x = x(1)$, respectively. Hence, for large changes, the willingness to pay may exaggerate the benefits. In a comprehensive analysis, the willingness to pay should be compared with MSC rather than MC in order to take account for the 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 are equal to the marginal willingness to pay, i.e., $p(2)$ in figure 1. This is the point at which the marginal willingness to pay equals the social marginal cost of abatement. The amount of energy saving is then $x(2)$.

In general, both the damage-cost approach and the willingness-to-pay approach therefore 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 (constant abatement costs), an estimate of the marginal damage cost would approximate the equilibrium price better than an estimate of the willingness to pay. The willingness-to-pay approach applies well if the demand for improved environmental quality is inelastic. A full analysis of the measures requires a macroeconomic model, which includes a specification of the marginal cost curve, and relations between economic activities and environmental effects of energy saving.

3. The model

This section describes how the damage cost approach and the willingness-to-pay approach can be identified within the context of a macroeconomic model. In order to focus on the methodological differences, we aggregate the model as much as possible. An extension is, however, straightforward in the sense that the model can be enlarged by sectors and commodities in accordance with standard macroeconomic modelling. We need to distinguish between three sectors of the economy to describe the main features of the approaches, a production sector, a health sector and households. The structure of the model is shown in figure 2, where square boxes indicate sectors, ovals indicate 'activities', and the arrows indicate 'deliveries'.

The production sector produces all commodities and services except for some health services. The production technology is assumed to exhibit constant elasticity of substitution (CES), where gross product in the production sector, x, is generated by input of labour, n_1, and of products from own sector, x_1,

$$x = \left(b + \beta_1 x_1^{\rho_1} + \beta_2 n_1^{\rho_1}\right)^{1/\rho_1}. \tag{1}$$

$1/(1-\rho_1)$ is the elasticity of substitution between input of commodities and services and labour, and β_1 and β_2 are the distribution parameters. The constant term, b, can be interpreted as the contribution to the gross output from real capital and natural resources. Real capital is affected by pollution through material damage, and the productivity of natural resources is affected by crop loss. Let B denote the total value of these stocks prior to the emission cuts, and denote the emission cuts by $\mathrm{d}m$. Then,

$$b = (1-\beta_1-\beta_2)\big[B(1+\xi\,\mathrm{d}m)\big]^{\rho_1}, \tag{2}$$

where ξ is the exposure–response coefficient for the total cost due to material damage and crop loss. Denote wages by w, and the buyer price of the output from the production sector by p, then the demand functions for x_1 and n_1 are

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

$$n_1 = (x^{\rho_1} - b)^{\frac{1}{\rho_1}} \left[\left(\frac{p^{\rho_1}}{\beta_1}\right)^{\frac{1}{\rho_1-1}} + \left(\frac{w^{\rho_1}}{\beta_2}\right)^{\frac{1}{\rho_1-1}}\right]^{-\frac{1}{\rho_1}} \left(\frac{w}{\beta_2}\right)^{\frac{1}{\rho_1-1}}. \tag{4}$$

The health sector focused in this study is limited to that part of the sector that treats people affected by pollution. The product of the health sector, z, is measured in terms of number of working days lost because of air pollution. Hence, the activity level in the health sector is dependent on the emissions. We assume that the sector exhibits a Cobb–Douglas technology

$$z = a n_2^{\mu} x_2^{1-\mu}. \tag{5}$$

The demand functions for labour and commodities and services are found by cost minimisation,

$$x_2 = \frac{z}{a}\left(\frac{w}{p}\right)^{\mu}\left(\frac{1-\mu}{\mu}\right)^{\mu}, \tag{6}$$

$$n_2 = \frac{z}{a}\left(\frac{p}{w}\right)^{1-\mu}\left(\frac{1-\mu}{\mu}\right)^{1-\mu}. \tag{7}$$

The total production cost in the health sector is one of the components of the social costs of health effects in the damage cost approach. In a macroeconomic context, it is often interpreted as a necessary public expenditure. In the model presented here, z will be determined indirectly by the demand for health standard in households.

The households derive utility from consumption of commodities and services, x_c, from the 'health status', y_c, and supply labour. Reductions in air pollution are assumed to improve the health status. We aim at expressing the demand for the health status and relate it to emission reductions. For many purposes, utility-yielding, non-market goods, such as

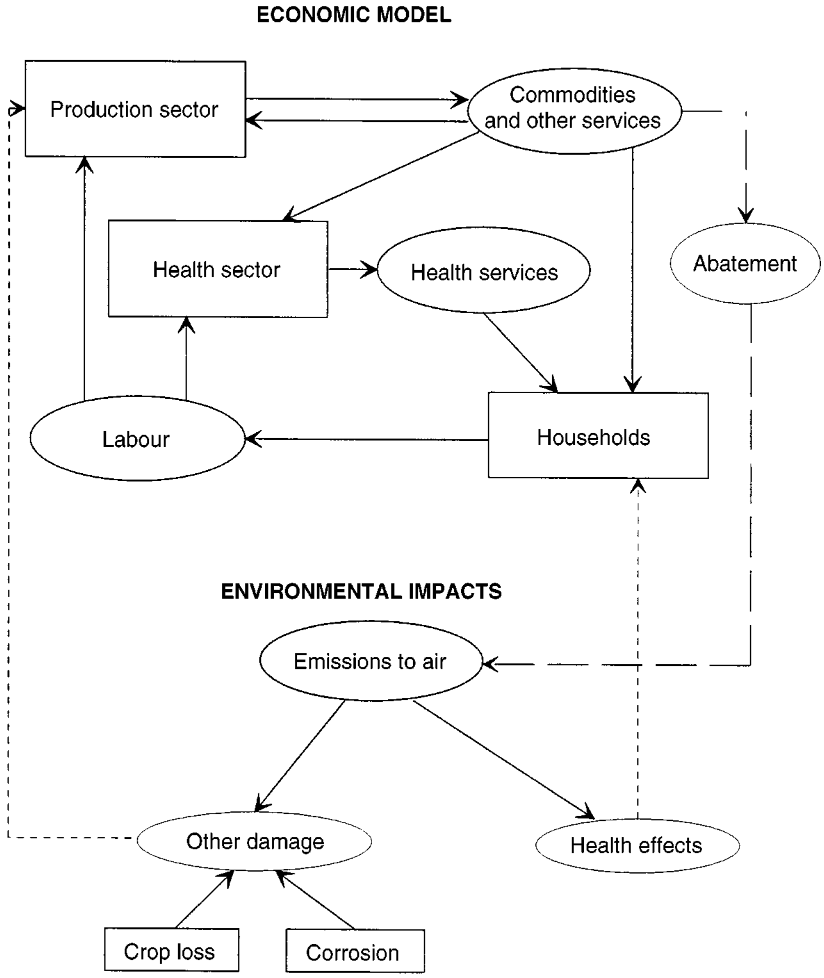

Figure 2. Structure of the macroeconomic model.

health, can be modelled by means of household production theory (see [11,16]), which is based on the assumption that certain goods and services 'produce' non-market goods. However, the input to generate the health status is not always paid by the households. In the model, the households indirectly demand emission cuts by their demand for health services. Health services are to some extent regarded as a public concern causing public expenditures. This raises two problems. First, the health status discussed here is not subject to individual choice, or at least only to a limited extent. We cannot, therefore, derive the demand for the non-market goods from the demand of the input, such as in the standard theory. Second, the budget constraint does not correspond to the 'true' budget of disposable income, but is extended to include measures to reduce emissions. In the model this problem is simplified by leaving all decisions about utility-yielding non-market goods, including public goods, to the households. In a more detailed model, one could alternatively add only public health expenditures, not really observed as expenditures for households, to the income in households. The households' budget constraint is therefore

$$R = px_c + qy_c, \tag{8}$$

where q is the price of health services and R is households' income including other public expenditures. We may therefore call this a virtual budget constraint. It implies that the 'optimal' level of emission reductions can be found under the assumption that the households, not public authorities, make decisions about the health sector.

To derive the demand for the health status we can use assessments of the willingness to pay for a marginal reduction in the frequencies of symptoms related to pollution, denoted r_j for symptom j. Although one may be sceptical to assessments of the willingness to pay, it is clear from the discussion in section 2 that it is impossible to estimate the value of environmental improvements properly without relating the estimates to the demand for the improvements. In this paper, we use the following approach: Assume that the "health status" measure is represented by an aggregate of improvements of n symptoms,

$$y_c = A \prod_{j=1}^{n} y_j^{\gamma_j}, \tag{9}$$

where the γ_i's ($\sum_{j=1}^{n} \gamma_j = 1$) are parameters expressing the 'importance' of each symptom, j. y_j is the effect on symptom j following an improvement in, e.g., air quality.

The households have a virtual budget for health improvements, $G = qy_c$, which is measured for instance by the total annual cost of measures to improve the health status, given the present status. Then, the total expenditure on the health services is

$$G = \sum_{j=1}^{n} r_j y_j. \tag{10}$$

The demand for reducing the frequency of symptom j is found by minimizing total expenditure at a given level of utility

$$y_j = G\frac{\gamma_j}{r_j}. \tag{11}$$

Surveys over the willingness to pay provide estimates of r_j. y_j relates to the present health status at the spot where the survey was carried out, but is not easily observed. If this knowledge is available, we can find γ_j, which characterises the utility of environmental improvements by a utility weight for each symptom. In principle, γ_j can be transferred from one country to another.[1] To apply the estimates in order to find the total willingness to pay, we need to know about G and y_j in the country to which the estimates are transferred. G is often assumed to be proportional to income in the two countries, while y_j is often assumed to be equal in two countries (see, e.g., [14,15]). Both assumptions are probably unrealistic. In the present model, G is determined endogenously by the demand for improved health, while we have had to assume y_j (the number of avoided cases due to the present health expenditures).

The macroeconomic model applied in this study is too aggregated to allow for a separation of different symptoms. Instead, we use the sum of the willingness to pay for five symptoms, based on bottom-up assessments, to model the demand for the health standard. To 'observe' the price of health, q, we use the willingness to pay estimate from the bottom-up assessments to determine G, and apply the assumptions about $y_1, \ldots, y_n$. In practice, R is the total private and public expenditure. In this simple model, we thereby leave all decisions about consumption to the households. Income is generated by the wage, w, that is

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

Assume that the utility exhibits constant elasticity of substitution (CES-utility). Then, households' demand for commodities and services and health improvements, respectively, can be written as

$$x_c = \left(\frac{p}{\alpha}\right)^{\frac{1}{\rho_2-1}} \frac{R}{\alpha^{\frac{1}{\rho_2-1}} p^{\frac{\rho_2}{\rho_2-1}} + (1-\alpha)^{\frac{1}{\rho_2-1}} p^{\frac{\rho_2}{\rho_2-1}}}, \tag{13}$$

$$y_c = \left(\frac{q}{1-\alpha}\right)^{\frac{1}{\rho_2-1}} \frac{R}{\alpha^{\frac{1}{\rho_2-1}} p^{\frac{\rho_2}{\rho_2-1}} + (1-\alpha)^{\frac{1}{\rho_2-1}} p^{\frac{\rho_2}{\rho_2-1}}}, \tag{14}$$

where α is the distribution parameter and $1/(1-\rho_2)$ is the elasticity of substitution between health and other commodities and services.

Emission control is introduced by specific measures, in this study described by the energy saving measures in the Energy Program. We assume that emissions are linearly dependent on the output in the production sector with coefficient ε, but can be reduced by implementation of energy saving measures at cost x_a. Hence, total emissions are

$$m = \varepsilon x - Hx_a^{\eta}, \tag{15}$$

where x_a^{η} is the inverted cost function (see figure 1) for the Energy Program, and H is a constant.

Most macroeconomic models used to study environmental policy represent emission control by means of a charge on emissions. The aim of a charge is to impose incentives to substitute the use of emission sources such as fossil fuels, to alternatives, e.g., renewable energy. Enabling studies of the impacts of charges is one of the main strengths of the macroeconomic approach. To do this properly, energy should at least be separated from other inputs to production. Such an extension of the model is straightforward, but avoided here because we want to focus on the Energy Program, not on the choice of alternative policy instruments.[2] Hence, this approach assumes that there are no other ways to reduce emissions in Hungary than by the Energy Program.

Equilibrium conditions can be imposed on the labour market, the product market and the market for emission reductions. Assume that there is a given, maximum supply of labour, $\bar{n}$. The actual supply, n, depends on the extent to which people are affected by poor air quality, measured by z. Thus,

$$\bar{n} - z = n_1 + n_2. \tag{16}$$

The loss of workdays due to air quality is estimated by the exposure–response relationship, based on assumptions of present excess effects of pollution

$$z = Sm^{\sigma}. \tag{17}$$

The simplifications made in this model means that $S = 1$. We have assumed also that $\sigma = 1$. In other words, we assume a linear relationship between energy use and health impacts. This assumption is rough, and applies only within a small interval. In the product market, we have

$$x = x_1 + x_2 + x_c + x_a. \tag{18}$$

Emission control refers to the given, initial level of emissions without control, εx. The households demand improved air quality which is directly related to emissions.

[1] We see from the demand function that in equilibrium γ_j can be defined as the budget share of symptom j in the health aggregate.

[2] For economies in transition, such as Hungary, the use of charges raises a number of questions about the effectiveness of market mechanisms and institutional requirements beyond the scope of this study (see, e.g., [24]). For a study of charges versus direct instruments in the control of greenhouse gas emissions, see [1].

Hence, the demand for improved air quality can be reinterpreted as demand for emission reductions. Then, in equilibrium we have

$$y_c = \varepsilon x - m. \tag{19}$$

The price of health services is the numeraire in the model, and according to Walras' law, the health market is in equilibrium.

4. Numeric assessments and results

The total annual saving of energy proposed in the Energy Program amounts to 63.7 PJ, or approximately 6% of total energy consumption in Hungary in 1995. The program specifies thirteen measures divided among different sectors of the economy. To estimate the effect on the emissions of local pollutants, Aunan et al. [6] assumed that the reduction of each primary pollutant is proportional to the amount of energy saving for the same measure within the same sector. Particles were considered as primary pollutants; for ozone, see [6].

The only information about costs is the total amount of 422 mill. USD for the entire program. We have assumed that this is the present value of the costs. The lack of documentation of this estimate has been criticised by OECD/IEA [20]. If the cost estimate is correct, the Energy Program is a very attractive alternative to improve Hungary's supply of energy, a major national task. However, the IEA doubts whether the program will be initiated at all, and points out that the costs may be severely underestimated, or the reductions in energy use are overestimated. Since no better alternatives are available, the 422 mill. USD are taken as the point of departure in this study. Because of the doubts whether the program will be implemented, we have disregarded the alternative price of energy and ask: If the reported costs of 422 mill. USD are regarded as the net cost of energy saving, should the program be implemented only because of its environmental benefits?

To establish a cost function for the Energy Program, a number of heroic assumption had to be made. Based on some vague information about how the 422 mill. USD were distributed among sectors, the thirteen measures of the program were collected into six groups. The duration of the measures in each group was chosen considering both how many years the technology would last before it has to be replaced, and how long time it would take before the measure would be implemented without the program. The discount rate was set to 10%. The annual cost of the 422 mill. USD is then 66.4 mill. USD. With a bottom-up approach, the annual environmental benefits of the Energy Program have to exceed this amount to be considered beneficial.

In a top-down model, the observed costs provide 'data' for estimating the parameters of the inverted cost function in equation (15). Table 1 describes the measures, and the annual savings and cost per unit of saved energy for each group of measures. The unit costs are calculated as the minimum constant price required over the lifetime of the measures to cover the total cost.

The estimated unit cost varies significantly among the groups of measures. Energy awareness, which is expected to contribute more than half of the savings in the Energy Program, exhibits very low unit costs. On the other extreme, thermal insulation and enhanced use of renewable energy have very high unit costs, even in terms of western European standards. The unit costs were applied to estimate a cost function for energy saving in Hungary. The cost function, and the 'observations' are displayed in figure 3. Because of the significant variations in the unit costs, the cost function becomes very steep at the end of the curve. In a macroeconomic model, we can see from figure 1 that the amount of energy saving thereby becomes very inelastic to changes in the willingness to pay. On the other hand, the total cost of energy saving may vary a

Table 1
Measures in the Energy Program.

Group of measure	Measure	Annual saving (PJ)	Cost (mill. USD/PJ)
Awareness	Improve energy awareness	34.5	0.02
New technology	In industry In agriculture Efficiency in energy production Efficiency in household utilities	7.2	0.04
Reduce loss	By efficiency in energy sector Transmission and distribution Co-generation in power plants	7.2	1.16
Energy management	Improve management in buildings	2.8	3.73
Insulation and renewables	Thermal insulation Renewable energy	2.5	7.40
Measures in transport	Co-operation in public transport Energy use in vehicles	9.5	1.93

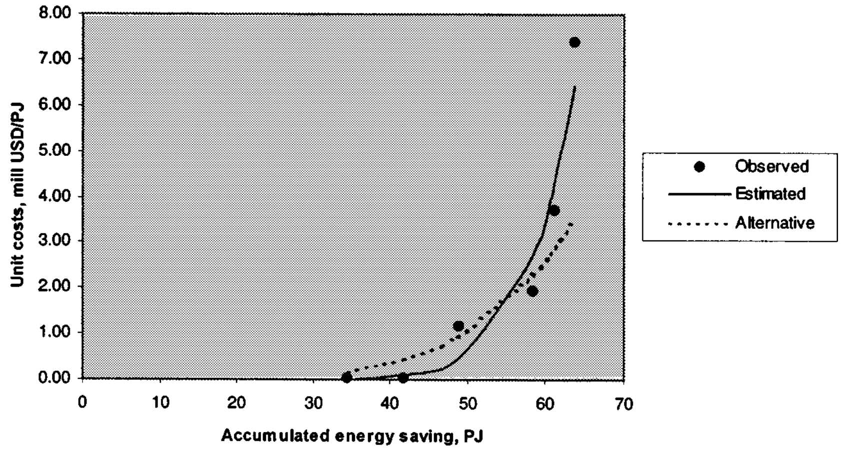

Figure 3. Cost curves for measures in the Energy Program.

Table 2
Estimated responses and effects in the labour market from implementation of the Energy Program.

Assumption/symptom	Expected response per year			Total increase in labour supply (man years)	
	Unit	Children	Adults	Patients	Health sector
Deaths[a]	no.	34	70	74	11
Lung cancer	no.	–	25	50	4
Acute respiratory symptom	Days/person	1	0.15	1 928	555
Chronic bronchitis	no.	14 040	16 520	3 100	248
Asthma	Days/person	–	2.4	433	48

[a] Deaths of people above 65 excluded.

lot. Recall, therefore, that the cost curve is based on very poor information. Less variation in the unit costs among measures would lead to a more gently sloping cost function, and hence a more flexible amount of optimal energy saving. To see the macroeconomic effects of such a shift, we have included a study of an alternative cost function, denoted 'alternative' in figure 3, in the macroeconomic analysis.

4.1. Damage costs

Aunan et al. [6] estimated the effects of the energy program on health, material damage and crop loss. The effect of tropospheric ozone on crops is substantial in Hungary. However, the concentrations of ozone are largely due to emissions outside Hungary. The effects in Hungary of measures taken in Hungary has therefore a limited impact on the crops. Aunan et al. [6] suggest an increase in the value of crops of 1–3 mill. USD per year as a result of the Energy Program.

The effects on material damage are larger. These are primarily due to reductions in the emissions of SO_2. Using statistics of building mass and materials and other studies in Europe, the reduction in the damages on buildings was estimated to 30–35 mill. USD in Budapest. Extrapolating this number to all cities in Hungary, the total amount of avoided damage could be estimated to 100 mill. USD per year.

The reduction in damages on materials and crops can be utilized to calibrate the damage parameter in the production function, i.e., ξ in equation (2). In the macroeconomic model, we assume that losses of crops and material damage cause a loss of stock resources that is proportional to the emissions.

Health is measured in terms of the effects on five end points, deaths, lung cancer, acute respiratory symptom, chronic bronchitis and asthma.[3] It is assumed that the reductions in concentrations are proportional to the emission cuts in the Energy Program. The health effects were calculated by exposure–response relationships from studies in other countries. The estimates cover cities only. The expected responses on health and on the labour market of the Energy Program are summarised in table 2.

Note that deaths, lung cancer and chronic bronchitis are given in number of cases, while acute respiratory symptoms and asthma are reported as days per person, since one person may experience several 'cases' during a year. To compare the effect on the different symptoms, we may calculate the damage cost, or the social cost reduction following the health effects of the program. These include the value of an increased labour force, partly from those subject to the symptoms, and partly from released labour in the health sector.

There are many studies of the costs of hospital admissions, but it is difficult to divide the costs into single cases of different symptoms. We have used Jacobsson and

[3] Pseudo croup was included as a sixth symptom, but was found to have a negligible effect in a macroeconomic context.

Lindgren's [13] study in Sweden. They provide estimates for the costs of the health sector (direct costs) and the costs related to the patients (indirect costs) for a number of symptoms, of which cancer and respiratory illness were closest to the symptoms studied here. They divide the direct costs into hospitalisation, consultations and medicines, while the indirect costs were divided into classes of seriousness. Based on rough assumptions about how the cost categories of direct costs are distributed among the different classes of seriousness, we calculated the cost of each symptom in terms of working days, assuming that non-labour costs (overhead) in the health sector are 1.5 times the costs of labour. The figures were then transferred to Hungarian standards by a comparison of wage levels.

A particular problem arises for the estimation of the costs of death. In bottom-up studies, it may be acceptable, as Jacobsson and Lindgren [13] do, to base estimates of the costs of death on the expected future earnings to account for some of the welfare loss of the person that dies. When assessing the damage costs in a macroeconomic context, however, the welfare effect should not be included in the damage costs, because this is taken into account of by the demand functions for health services. There are in principle no economic effects of a death apart from the potential loss of labour and the avoided cost to the health system the year the person dies. The damage cost of deaths is therefore calculated on an annual basis.

The impacts on the labour market shown in table 2 include parents' absence from work due to illness of their children. The figures show that the effects on acute respiratory symptoms and chronic bronchitis are the most significant contributors to the damage costs since they affect more people. In particular, the benefit of a reduction of chronic bronchitis is large, because this disease is assumed to have a considerable impact on the productivity of people.

The annual reduction in costs related to health was estimated to 40.7 mill. USD per year. Together with the reduction of material damage and crop loss, the total reduction in damage costs from pollution is then approximately 142 mill. USD per year, which compares with the annual costs of 66.4 mill. USD. In other words, the damage cost approach turns out with a positive net benefit of the Energy Program, if the assumptions about the costs are correct. Since this approach gives the minimum social benefit of the program, one might claim that this estimate provides a strong argument for implementation of the program. One could, however, question this conclusion for two reasons. First, all exposure–response relationships have large uncertainties. Estimates based on epidemiological data, such as the health effects calculated here, exhibit wide ranges of uncertainties. For instance, Aunan et al. [6] estimated a 95% confidence interval to lie between 6 and 126 infant deaths, and between 360 and 76 550 number of cases for chronic bronchitis in children. A Monte Carlo simulation of the reduction in damage costs of health effects, using the intervals for the responses of each symptom in Aunan et al. [6], resulted in a 90% confidence interval between 15 and 130 mill. USD. In addition to the uncertainties about the parameters of the exposure–response relationship, there is uncertainty about the transformation of data from one country to another. Systematic bias may be due for instance to differences in smoking habits among countries.

Second, the costs of measures and damages reported above apply to the whole program. However, the costs vary greatly among the different measures. Implementation should in principle be restricted to the part of each measure within the Energy Program that can be implemented at lower marginal costs than marginal benefits. If we assume that marginal benefits equal average benefits, the marginal benefit in terms of damage costs is 2.2 mill. USD/PJ. If we further assume that each measure will either be fully implemented or not at all, the measures grouped under energy management and insulation and renewables should then be taken out of the program. This reduces the saving potential of the Energy Program to 57.9 PJ.

4.2. Willingness to pay

The major objection to using damage costs as a proxy for the benefits is that it disregards the welfare aspect of being healthy. To include this, we have to find a measure directly related to the welfare of the health status. In this study, we have chosen the willingness to pay assessed by contingent valuation methods. Assessments of the willingness to pay has received a lot of attention in recent years. Besides the applicability in practical policy making, one of the main advantages of this approach is that it provides numerical assessments of the valuation of non-market issues. The main problem is that the approach is based on the perceptions of those who are interviewed, while market prices can be revealed from behaviour. Hence, we do not know exactly how to interpret the answers. The appropriateness of surveys of the willingness to pay has been intensively debated (see, e.g., [5,12,17]), and there is a lot of scepticism to the method. We will not contribute to the discussion here, but rather emphasise the need for numerical assessments when trying to estimate the value of the environment. Hence, we confine ourselves to assume that it is possible to obtain reliable estimates.

To our knowledge there are no surveys of the willingness to pay for improved air quality available in Hungary. Hence, we have to draw on studies from other countries. Attempts to study the validity of such transfers have given different conclusions. However, Alberini et al. [3] do not find evidence to discard the hypothesis that the utility functions applied to environmental quality is the same when comparing the US and Taiwan.

The model presented above illustrates some reasons why it is problematic to compare willingness to pay estimates across countries. One is that one needs to know the state of the environment referred to both in the country where the survey is carried out, and in the one to which the estimates are transferred. In the case examined here, we need to make some assumptions about the number of excess symptoms

Table 3
Willingness to pay for the health effects in the Energy Program under alternative assumptions.

Symptom	WTP per unit[a,b] (USD)	Estimated value of benefits from the Energy Program (mill. USD per year)		
		Income level US	Income level Hungary[c]	
			Same as in the US	Three times more than in the US
Deaths	450 000	468	75	125
Lung cancer	300 000	75	12	20
Acute respiratory symptom[d]	98	198	32	53
Chronic bronchitis	240 000	1 650	264	441
Asthma	36	23	4	6
Total	–	2 414	386	645

[a] See table 2 for reference to units.
[b] EPA (1995). Deaths of people above 65 excluded.
[c] No. of cases related to pollution in Hungary relative to the US.
[d] Adjusted to take account for different degrees of seriousness.

due to air pollution at present. The exposure–response relationships provide this information for the country for which the relationships were estimated, which is mainly the US. However, we did not have similar data for Hungary. Instead we have calculated how sensitive the estimate for the willingness to pay for the Energy Program is to alternative assumptions about the relation between US and Hungarian health standards.

A second assumption that has to be made is the level of expenditures on health services related to the five symptoms in the two countries. Here also the data for the US are insufficient. In the calculations presented below, we have assumed that these expenditures are proportional to income. This assumption is, however, not fully consistent with the model, since the expenditures are determined endogenously. To check for this bias, we may see how sensitive the expenditures on health are to alternative estimates of the willingness to pay in the macroeconomic model.

Table 3 shows the alternative estimates of the willingness to pay for the Energy Program, based on surveys in the US. The level of expenditure in Hungary, assumed proportional to income, is 16% of the US income. We have calculated the willingness to pay under two alternative assumptions about the environmental quality, in this case the pollution related health status. In one alternative, we assume that the same fraction of people is affected by air pollution in Hungary and the US. This is a quite usual assumption, but obviously not realistic (see, e.g., [15]). In the other alternative, it is assumed that three times as many people are affected by pollution in Hungary as in the US.

Using results from willingness-to-pay studies in the US directly, the willingness to pay for a similar improvement as those expected by the Energy Program in the US would be more the 2.4 bill. USD per year. Because of the lower income level in Hungary, the willingness to pay drops to 386 mill. USD, if we disregard the different environmental standards in the two countries. The estimate is, however, sensitive to this assumption. If the fraction of people affected by air pollution is three times as high as in the US, the willingness to pay increases to 645 mill. USD. What is surprising is that chronic bronchitis accounts for nearly 70% of this amount.[4] This indicates also that the willingness to pay for avoiding chronic diseases is very high compared with the willingness to pay for avoiding short term illnesses.

According to these figures, the Energy Program is indeed beneficial. Again, there might be some modifications to the conclusion. One is the assumption that it is possible to transfer willingness to pay estimates between the US and Hungary at all. This assumes that the utility function is similar in the two countries. Technically speaking, it means that γ_i in equation (9) is similar for all i in the two countries. This is clearly a strong assumption, even though it may be difficult to discard the hypothesis that they are similar.

The other reason to be sceptic about the figures in table 3 was mentioned in section 2, when presenting the properties of willingness to pay assessments in a macroeconomic context. From figure 1, we know that the estimates are likely to be excessive, because the willingness to pay drops as the environment improves due to the implementation of the Energy Program. Instead of measuring the area under the demand curve, we have measured the shaded square in the figure. To approach equilibrium quantities and values, we need to solve the macroeconomic model numerically.

4.3. A macroeconomic analysis

Macroeconomic analyses usually calculate the cost of pollution control as the social cost of reducing the emissions, measured for example by the associated reduction in GDP. This may be compared with an estimate of the benefits. In the present model, estimates of the costs and the benefits are obtained simultaneously. The model determines how much of the Energy Program that will be implemented in equilibrium if the benefits are taken into account. To compare the Hungarian economy with and

[4] Note that deaths of people older than 65 years are excluded. In [6], the benefits of avoided deaths of people above 65 contribute to approximately 40% of the total benefits.

Table 4
Changes from the point of reference to equilibrium conditions after implementation of the Energy Program under alternative assumptions about abatement costs and willingness to pay.

	Unit	Base case	Alternative costs	Lower WTP
Increase, GDP	Mill. USD	381	382	257
Increase, consumption of commodities, and services	Mill. USD	−99	−110	−3
Increase in labour supply	Man years	5 255	5 898	4 951
Total abatement costs	Mill. USD	44	78	18
Energy conservation	PJ	64.5	72.0	59.6
Emission reductions	Pct	6.9	7.8	6.6

without the Energy Program, we compare two states of the economy. The point of reference is the initial situation where the commodity and service market and the labour market are in equilibrium, and the Energy Program is *not* implemented. The activity in the health market is then exogenously determined. This means that there is no relation between the activity in the health sector and households' demand for emission reductions in the reference case. In the alternatives there is equilibrium in the health market as well, and the damage on materials and crops is taken explicitly into account by the decision makers.

We consider three cases where the relations between environmental impacts are included. In the base case the Energy Program, as described by the cost curves in figure 3, is implemented. The willingness to pay in the base case corresponds to the alternative where the health status in Hungary is three times as bad as in the US (table 3), i.e., 645 mill. USD. In the "alternative costs" case there is less variation in the unit costs among measures in the Energy Program. The cost curve then corresponds to the dashed curve, denoted "alternative", in figure 3. In the third case, "Lower WTP", the willingness to pay for the Energy Program is assumed to be half as high as in the base case, i.e., 323 mill. USD. This is slightly lower than the estimated willingness to pay given that the environmental status in Hungary is equal to the environmental status in the US (see table 3).

Table 4 presents the main results in terms of deviation from the reference point. According to the calculations an implementation of the Energy Program will enhance GDP by 381 mill. USD, or 0.4% compared to the point of reference. A part of this increase is due to a reduction in material damage. Although small 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.

In the *base case* a reduction in households' consumption of ordinary commodities and services of 0.3% is compensated by a better health status, which amounts to more than 5000 man years. As a consequence of the improved health status, the activity in the health sector is reduced. The increase in the supply of labour is channelled to the commodity and service sector, which together with reduced material damage explains the increase in output. The sector needs to use more of its own output as input to manage the new level of output. The remaining part of the reduction in households' consumption is spent on the measures in the Energy Program.

Total abatement costs in equilibrium are calculated to be 44 mill. USD per year. Recall that this figure cannot be compared directly with the observed costs, since it is based on the estimated cost function. The abatement costs allow for 64.5 PJ energy to be conserved. In other words, $x(2)$ in figure 3 is 64.5. We note that total abatement costs in equilibrium amounts to only 6% of the estimated willingness to pay in the bottom-up approach.

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 add nearly 5 mill. tons of CO_2-emissions to their own targets. The results reported in table 4 show, however, that the output of commodities and services increases. As a consequence, 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 leads to a 'leakage' of the emission cuts at about 7%, 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.

In the *alternative cost case* the least costly measures become more expensive, while the most costly measures cost less than in the base case. All measures beyond those explicitly included in the Energy Program are therefore expected to be less costly than in the base case. Since the marginal cost curve is more gently sloping, the amount of energy conservation increases to 72 PJ, or by 7.5 PJ. The total cost of abatement in optimum is 78 mill. USD per year. This leads to a better health status and an increase in the labour force by approximately 650 man years, compared to the base case.

The increase in total abatement costs leads, however, to a relative shortage of commodities and services compared with labour. At the same time the supply of labour increases. This is partly because less people are affected by

Table 5
Alternative assessments of energy saving potential, costs and/or environmental benefits of the Energy Program.

	Energy conservation (PJ)		Total value of assessment[a]	Marginal value	
	Energy Program	National reduction in energy use	(mill. USD)	(mill. USD/PJ)	(cents/kWh)
Abatement costs	63.7	63.7	66.4	7.4	2.7
Bottom-up					
Damage costs	63.7	63.7	141.5[b]	2.2	0.8
Willingness to pay	63.7	63.7	645.0[b]	10.1	3.6
Top-down					
Basis	64.5	60.0	43.5	7.3	2.6
Alternative cost function	72.0	67.5	78.3	6.4	2.3
Lower WTP	59.6	56.5	18.1	3.3	1.2

[a] See text for explanation.
[b] The estimate value of damage costs includes material damage and damage on crops. The willingness to pay approach assumes willingness to pay only for improvement of the health standard.

pollution related diseases. As a result, the price of health services is lower than in the base case, and the activity in the health sector goes down. The increase in labour supply affects the wages negatively, and consumption is slightly lower than in the base case. The output from the commodity and service sector is approximately the same as in the base case, but the sector has to some extent substituted the input of commodities and services by labour. This reduction in the use of products from own sector enables an increase in the expenditures for the Energy Program. The cost of the program in the alternative cost case is nearly twice as high as in the base case.

In short, more slowly increasing marginal abatement costs lead to a better health status and less consumption. 72 PJ are saved by the Energy Program. This leads to nearly a doubling of the total cost of the program. Since GDP is practically speaking unaltered, the better health status can be regarded as a compensation for the increase in the expenditures for energy conservation.

In the case of *lower willingness to pay*, an increase in the demand for consumer goods leads to approximately the same consumption level as in the point of reference. Expenditures on health constitute about 0.7% of the households' budget, compared with 1.6% in the base case. Recall that this makes the assumptions used when transferring estimates of the willingness to pay between countries questionable, since the transfer assumed that the expenditures to health services were unaffected by a shift in the total willingness to pay. The reduction of the budget share is mainly due to a significantly lower shadow price of emission cuts in this case. Compared with the point of reference where the Energy Program is not implemented, the lower willingness to pay case thus implies a significant improvement in the health status, amounting to 4950 man years. This improvement may be carried through without any notable reduction in the consumption level. The amount of energy saving is 59.6 PJ in this case. Compared to the base case, the activity in the health sector is slightly increased, due to higher emissions, although higher emissions compared to the base case are counteracted to some extent by a lower output of commodities and services.

The total cost of abatement measures in this alternative is only 18.1 mill. USD. In other words, 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. Due to the lower output of the commodity and service sector, the difference between energy saved directly in the Energy Program and total national reduction in energy use, the "leakage", has decreased in this case compared to the base case.

A comparison of the different estimates of marginal values is shown in table 5. The bottom-up assessments all assume that 63.7 PJ of energy is conserved, that is, the potential given by the Hungarian authorities in their presentation of the Energy Program. For the top-down assessments, the estimated amounts of conserved energy vary. The two columns show the energy directly saved by the implementation of the Energy Program, and the national savings, which include the effect of "leakage". The total value of the assessments refers to the estimated cost of the Energy Program in the abatement cost row. This is also the case for the top-down rows. For the bottom-up approaches the column shows estimated benefits. The marginal values of the program are assumed to be equal to the average unit cost of conserved energy for the most expensive measure, insulation and renewables. These correspond to the negative shift in the cost curve and the point p(WTP) in figure 1, respectively. The willingness to pay estimate implies that the value of reduced emissions amounts to 3.6 cents/kWh, which is about one third of the user price for electricity in western Europe. This is hardly realistic. The marginal value of better air quality according to the damage cost estimate, 0.8 cents/kWh, seems to be more acceptable.

All the values calculated by the top-down model lie between these two estimates. The value of energy saving in the base case is relatively high, 2.6 cents/kWh. This is due to the steep cost curve for energy saving at the margin, which reflects strong limitations for finding new ways to save energy. Also when the cost function is more flexible, the estimated value of a better environment is substantially lower, 5–10% of the 'normal' price of electricity in Western Europe. Nevertheless, this is still high in Hungary, where the households pay a very low price for energy. Hence it may indicate that it is unrealistic to transfer willingness to pay estimates from the US to Hungary. Lower willingness to pay reduces the value of improved air quality to 1.2 cents per kWh, which is probably more realistic.

5. Conclusions

According to the figures in this paper, estimates of the value of environmental quality are strongly dependent on the approach. In particular, estimates based on bottom-up studies may differ considerably according to the method used. 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 macroeconomic 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. To do this, a number of assumptions had to be made. 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 could be estimated with more reliable information than provided for this study. 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.

It should be emphasised that this study focuses on the importance of approaches to the valuation of environmental qualities, and that the numerical assessments are meant mainly as illustrations. However, the Energy Program was originally presented as a means to reduce greenhouse gas emissions in order to make Hungary keep track with the expected commitments in an international treaty on greenhouse gas emissions. The results show 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 'secondary', 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 may be a subject for future research.

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