THE VALUATION OF HEALTH IMPACTS IN DEVELOPING COUNTRIES

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Abstract

Valuing the health impacts in money terms can provide an important aid to the decision-maker in setting priorities for investments that improve the quality of life. In many developing countries, however, a shortage of data often make the estimation of these impacts not practicable in the time available. This paper, after providing the conceptual basis for these valuations, discusses how data and functions from other sources can be transferred to the site and country in question with specific examples for developing countries.

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1 Introduction

Probably the most important benefit of development is improvement in the quality of life. This improvement is measured to a large extent through increased life expectancy and reduced morbidity – reduced incidence of illness. While it is true that *general development,* which results in better nutrition and improvements in housing, water supply and sanitation, will improve the quality of life, it is not true that all investments in these areas are equally desirable. Nor are these the only investments that Impact on the quality of life. Others include measures to reduce air pollution, investments in public and private health provision and education for women.

In all these areas of policy, valuing the health impacts in money terms can provide an important aid to the decision-maker. How much should we spend on controlling emissions from vehicles, and how much can we justify in investments in improving drinking water supply. If there is a budget constraint and we have to choose between these options, which is the more valuable?

This paper is devoted to a discussion of the values to be attached to health impacts in developing countries. It is structured as follows. Section 2 provides the conceptual basis for the valuation. Section 3 discusses how we can overcome a shortage of data and information about these values. In many developing countries, there is a shortage of data and it is not practicable to collect what is required in the time available. The paper discusses how data from other sources can be transferred to the site and country In question. Section 4 focuses on the valuation of mortality effects -- effects that reduce the risk of death. Section 5 deals with the valuation of morbidity effects. Section 6 concludes the paper.

1 Conceptual Basis for Valuation of Health Impacts

2.1 Willingness to Pay and Willingness to Accept

Over the last 25 years or so, a number of techniques have been developed for estimating external environmental effects. A survey of these may be found in Markandya and Richardson (1993), with examples of their application in non-OECD countries in Winpenny (1996). In this section we describe the concepts underlying the valuation and the review the debate surrounding the use of such a system of valuation.

The underlying principle in monetary valuation is to obtain the willingness to pay (WTP) of the affected individual to avoid the negative impact, or the willingness to accept (WTA) payment as compensation if a negative impact takes place. The rationale is that values should be based on individual preferences, which are translated into money terms through individual WTP and WTA.

Once the impacts have been identified in physical terms, they can be valued using market prices, where the things impacted (crops, materials etc.) have a market price, although even in this simple case there are problems and issues that arise, which are discussed further below. For a wide range of impacts, however, such as increased risk of death or loss of recreational values, there are no direct market prices that can be used. There are three techniques that are widely used for the valuation of such. One is to elicit the WTP or WTA by direct questionnaire. This is termed the contingent valuation method and has been developed into a sophisticated procedure for valuing a number of environmental impacts. Another is to look at the WTP as expressed in related markets. Frequently environmental effects are reflected in property values. Thus an increase in noise or a reduction in visibility will "show up" in reductions in the value of properties affected by the changes.

This approach is called the hedonic price method and is widely used for noise and aesthetic effects.

Where the individuals undertake expenditures to benefit from a facility such as a park or a fishing area one can determine their WTP through the expenditures on the recreational activity, including costs of travel to the park, any fees paid etc. Economists have developed quite sophisticated procedures for estimating the values of changes in environmental facilities using such data. This method is known as the travel cost method and is particularly useful for valuing recreational impacts.

2.2 Categories of Value

The WTP/WTA numbers can be expressed for a number of categories of value. The most important distinction is between values arising from the use of the environment by the individual and values that arise even when there is no identifiable use made of that environment. These are called **use values** and **non-use values** respectively. Non-use values are also sometimes referred to as **existence values.**

Within the category of use values there are many different categories. **Direct use values** arise when an individual makes use of the environment (e.g. s/he breathes the air) and derives a loss of welfare if that environment is polluted. **Indirect use values** arise when an individual's welfare is affected by what happens to another individual. For example, if you feel a loss of welfare as a result of the death or illness of a friend or relation, resulting from increased levels of air pollution, then this loss of welfare translates into a cost through your WTP. It can and has been measured in limited cases and is referred to as an **altruistic value** (see later in this section for details). Both direct and indirect use values have a time dimension; an environmental change today can result in such values now and in the future.

Another category of use value that is potentially important is that of **option value**. This arises when an action taken now can result in a change in the supply or availability of some environmental good in the future. For example, flooding a region to impound water for a hydro project would result in that area not being available for hiking. A person might have a WTP for the option to use that hiking area, even if s/he was not sure that it would be used. This WTP is the sum of the expected gain in welfare from the use of the area, plus a certain gain in welfare from the knowledge that s/he could use it even if it is not actually used. The latter is referred to as the option value. The literature on environmental valuation shows that, in certain cases the option value will be positive but in general it is not an important category of value (see Freeman, 1991). There are very few estimates of such values, and in the context of most health valuation studies the issue is not likely to be important.

The last category of value is non-use value. This is a controversial category, although values deriving from the existence of a pristine environment are real enough, even for those who never make any use of it. In some respects what constitutes 'use' and what constitutes 'non-use' is not clear. If someone sees a programme about a wilderness area but never visits it, that represents a use value, however indirect. Pure non-use value must not involve any welfare from any sensory experience related to the item being valued. In fact some environmentalists argue that such non-use or existence values are unrelated to human appreciation or otherwise of the environment, but are embedded in, or intrinsic to, the things being valued. However, that is not the position taken in this paper. The basis of valuation remains therefore an anthropocentric one that, however, does not imply an anti-environmentalist stance.

The difficulty in defining non-use values extends, not unnaturally, to measuring them. The only method available for this category is that of the questionnaire approach, or contingent valuation. This method has been tested and improved extensively in the past 20 years, and the general consensus is that the technique works effectively where 'market conditions' of exchange can be simulated effectively and where the respondent has considerable familiarity with the item being valued (Arrow et al, 1993). For most categories of non-use value this is simply not the case. Hence, for the present, non-use values are extremely difficult to value with any accuracy.

2.3 Issues Arising In the Use of Monetary Values

Thus the basic philosophy underlying the valuation is based on individual preferences, which are expressed through the willingness to pay (WTP) for something that improves individual welfare, and willingness to accept payment (WTA) for something which reduces individual welfare. The total value of environmental impacts is taken as the sum of the WTP or WTA of the individuals comprising it. Thus no special weight is given to any particular group. This approach contrasts, for example, with that of values based on expert opinion, or values based on the costs of making good any damage done to the environment by an investment programme. Such mitigation costs will only provide a valid measure of cost if society is collectively willing to pay for the mitigation, rather than suffer the damage. In such cases mitigation based estimates can provide important values, and have in fact been used in the study in selected areas. However, the validity of that use is dependent on the assumption that society is willing to pay for the mitigation.

Although the valuation of environmental impacts using money values is widespread and growing, there are still many people who find the idea strange at best and distasteful and unacceptable at worst. Given the central role being played by monetary valuation in this exercise, a justification of the method is warranted.

One objection often voiced in the use of WTP is that it is 'income constrained'. Since you cannot pay what you do not have, a poorer person's WTP is less than that of a richer person, other things being equal. This occurs most forcefully in connection with the valuation of a statistical life (VOSL) (which is discussed in greater detail in Section 3.X) where the WTP to avoid an increase in the risk of death is measured in terms of a VOSL. In general one would expect the VOSL for a poor person to be less than that of a rich person. But this is no more or less objectionable than saying that a rich person can and does spend more on health protection than a poor person; or that individuals of higher social status and wealth live longer on average than person of lower status; or that better neighbourhoods will spend more on environmental protection than poorer neighbourhoods. The basic inequalities in society result in different values being put on the environment by different people. One may object to these inequalities, and make a strong case to change them but, as long as they are there, one has to accept the consequences. One could argue, for example, that increased expenditure on high technology medicine in Europe is unethical, even though the citizens of that region have a WTP that justifies such expenditures, because the same expenditure on preventative medicine in a poor developing country would save more lives. However, society does not accept such an argument, taking the view that most decisions about allocation of resources are predicated on the existing inequality of income and wealth, both between and within societies.

In conclusion, we can see that, although there are some objections to the use of WTP/WTA as a basis of valuing extemalities, it is by far the most intellectually defensible basis for valuation in a liberal society. Policy-makers may wish to pay attention to other aspects of extemalities, such as how many people are affected, how many of them are 'poor' etc. It is only right and proper that they should take account of such factors. The values associated with the extemality, calculated within the above framework are therefore only part of the information that will eventually determine the selected policy. But the valuation of the extemality, in money terms, needs a rigorous basis and the WTP/WTA approach provides that basis.

3 Transferability of Benefit Estimates

3.1 Introduction

The environmental damages associated with a particular investment will depend on the precise details of that investment: location, population-impacted etc. Clearly, it would be infeasible to estimate all environmental damages for each programme *ab initio*. Much of the work required is extremely time consuming and expensive, making the transfer of estimates from one study to another an important part of the exercise. The difficult issue is to know when a damage estimate is transferable and what modifications, if any, need to be made before it can be used in its new context.

3.2 Benefit Transfer

Benefit transfer is "an application of monetary values from a particular valuation study to an alternative or secondary policy decision setting, often in another geographic area than the one where the original study was performed" (Navrud (1994)). There are three main biases inherent in transferring benefits to other areas:

a) Original data sets vary from those in the place of application, and the problems inherent in non-market valuation methods are magnified if transferring to another area.

b) Monetary estimates are often stated in units other than the impacts. For example, in the case of damage by acidic deposition to freshwater fisheries, dose response functions may estimate mortality (reduced fish populations) while benefit estimates are based on behavioural changes (reduced angling days). The linkage between these two units must be established to enable damage estimation.

c) Studies often estimate benefits in average, non-marginal terms and do not use methods that are transferable in terms of site, region and population characteristics.

Benefit transfer application can be based on: (a) expert opinion, or (b) meta analysis. Expert opinion looks at the reasonableness involved in making the transfer and in determining what modifications or proxies are needed to make the transfer more accurate. In many cases expert opinion has been resorted to in making the benefit transfer during the US/EC study. In general the more 'conditional' the original data estimates (e.g. damages per person, per unit of dispersed pollution, for a given age distribution) the better the benefit transfer will be. In one particular case (that of recreational benefits) an attempt was made to check on the accuracy of a benefit transfer by comparing the transferred damage estimate with that obtained by a direct study of the costs. The finding there was not encouraging in that the two figures varied by a wide margin.

3.3 Meta Analysis

Where several studies, reporting a similar final estimate of environmental damage, exist, and where there are significant differences between them in terms of the background variables, a procedure known as meta-analysis has been developed to transfer the results from one study across to other applications. What such an analysis does is to take the estimated damages from a range of studies of, for example, coal fired plants and see how they vary systematically, according to affected population, building areas, crops, level of income of the population, etc. The analysis is carried out using econometric techniques, which yield estimates of the responsiveness of damages to the various factors that render them more transferable across situations. This can then be used to derive a simple formula relating environmental costs to per capita income, which could then be employed to calculate damages in countries where no relevant studies were available.

Estimates of damages based on meta-analysis have been provided in a formal sense in two studies carried out in the US on recreation demand (Smith and Kaoru (1990)), Walsh, Johnson and McKean (1989)), and on air pollution (Smith and Huang (1991)). The results in the recreation studies indicate that, as one would expect, the nature of the site is significant for the WTP attached to a visit, as are the costs of substitutes and the opportunity cost of time. Choice of functional form in the estimating equations also appears to play a part. In the air pollution study referred to above, it was found that damages per unit of concentration vary inversely with the average price of property in the study (the higher the price the lower the unit value of damage). If correct, it would enable an adjustment to the estimated value to be made on the basis of the average prices of properties in the area being investigated. However, the authors are cautious about the validity of the estimates obtained.

A formal meta-analysis is difficult to carry out. However, sometimes 'expert' adjustments can provide an informal meta analysis. For example, adjusting estimates of damages for size of population to obtain a per capita estimate and transferring that to the new study implicitly assumes that damages are proportional to population. Such adjustments are frequently made.

3.4 Adjusting for WTP on the basis of *per capita* income

An important rule of thumb that has been used in arriving at values of WTP for countries where there are no studies is to take the WTP estimates from the EC, US, or other OECD country and adjust it for the different in real *per capita* income. This was suggested by Markandya (1994), and has been applied by Krupnick et al. (1996) and others, including the World Bank studies on the valuation of air quality.

In making the adjustment the underlying assumption is that there is an 'elasticity' of WTP with respect to real income. The elasticity measures the percentage by which the WTP for a particular benefit declines for each percentage fall in the real income of the person concerned. One assumption that has been commonly used is that the elasticity is one. Another value, taken from Mitchell and Carson's work in the US is for an elasticity of 0.35 (Mitchell and Carson, 1986). In the case of Brazil, for example, the *per capita* income, adjusted for purchasing power was $5,400 in 1994 (calculated in 1994 prices). In the same year, the real *per capita* income of the US was $25,880. If we take an elasticity of one, the implication is that an impact, which has a value of $1 in the US would have a value of 21 cents in Brazil. If the elasticity is 0.35, the same impact will have a value of 58 cents in Brazil. In some of the key values reported below, both sets of estimates have been taken, and indeed similar estimates have been made for a range of countries. Although crude, this method is considered to provide a rough guide to health damage values that can be used for many investment decisions.

3.5 Conclusions on benefit transfer

Transferability depends on being able to use a large body of data from different studies and estimating the systematic factors that would result in variations in the estimates. In most cases the range of studies available are few. More can be done to carry out meta-analysis of the type indicated, but it will take time. The best practice in the meantime is to use estimates from sources as close to the one in which they are being applied and adjust them for differences in underlying variables where that is possible. Often the most important obstacle to systematic benefit transfer, however, is a lack of documentation in the existing valuation studies.

From the environmental damage-energy source linkages identified above, one can identify an increasing order of difficulty (in terms of modifications that have to be made) with which estimates can be transferred from the original study to the situation in which they are to be used;

a) The most easily transferred data is the dose-response function itself, relating environmental impacts adjusted for population. Thus numbers in the form: 0.8×10^{-6} excess deaths per $\mu g/m^3$ would be transferable across studies as long as adjustments to the other variables in the dose-response function were made (e.g., relative humidity, population at risk etc.). The additional local information that is required to use such data is simply local market conditions, costs and prices.

b) The next ones in order of difficulty are monetary estimates of damages per unit of pollutant by concentration. Results are reported, e.g., in ECU/$\mu g\ m^{-3}$, or in \$/km/person of lost visibility. Estimates may vary according to population affected, in which case an analysis of such variations would be desirable. Other socio-economic variables that would be of relevance are income levels of the affected population, age, background environmental variables such as rainfall etc., and socio-economic variables such as medical services and how they are paid for. If enough studies are available, a meta-analysis can be performed (see below), in which the mean estimated value is regressed against these variables. Then the relevant adjustment to the estimates is made, given the local values of the explanatory variables. No additional local variables should be required. In other cases the income elasticity may be used, as was done in the previous section.

c) Similar to (b) above are estimates of monetary damages in terms of emissions or units of energy produced. In such cases one needs all the information listed above, plus details of how the emissions or energy units relate to the concentrations or whatever impacts are responsible for the damages. For example, damages may be quoted as \$x/kWh for coal. The relevance of this estimate to a different situation will depend on how the kWh is related to emissions and how the emissions are converted into concentrations in the area where the impacts were measured, plus the variables with which the relationship between concentrations and damages vary. Thus most work will have to be done in these cases and, for many purposes it is unlikely that such estimates can be used at all.

It is important to note that national boundaries themselves are not of any relevance in transferring estimates, except that there may be cultural differences that will influence factors such as frequency with which a person visits a doctor, or how he perceives a loss of visibility. In this sense there is no reason why a Brazilian project should not draw on the US and other studies, or transfer estimates from one country to another within Europe, as long as the above consideration is taken into account.

4 Valuation of Health Impacts — Mortality

The final subsection here deals with the most important of the direct impacts of air pollutants – those on human health. These are divided into mortality effects and morbidity effects.

The mortality approach in the valuation literature has been mainly based on the estimation of the willingness to pay for a change in the risk of death. This is converted into the 'value of a statistical life' (VSL) by dividing the WTP by the change in risk. So, for example, if the estimated WTP is $100 for a reduction in the risk of death of 1/10000, the value of a statistical life is estimated at 100*10000, which equals one $1 million. This way of conceptualising the willingness to pay for a change in the risk of death has many assumptions, primary among them being the 'linearity' between risk and payment. For example, a risk of death of 1/1000 would then be valued at $1mn/1000, or $1000 using the VSL approach. Within a small range of the risk of death at which the VSL is established this may not be a bad assumption, but it is clearly indefensible for risk levels very different from the one used in obtaining the original estimate.

Estimates of the WTP for a reduction in risk or the WTA of an increase in risk have been made by three methods. First, there are studies that look at the increased compensation individuals need, other things being equal, to work in occupations where the risk of death at work is higher. This provides an estimate of the WTA. Second, there are studies based on the CVM method, where individuals are questioned about their WTP and WTA for measures that reduce the risk of death from certain activities (e.g., driving); or their WTA for measures that, conceivably, increase it (e.g., increased road traffic in a given area). Third, researchers have looked at actual voluntary expenditures on items that reduce death risk from certain activities, such as cigarette smoking, or purchasing air bags for cars.

In the environmental economics literature, mortality impacts are valued by multiplying the change in risk of death by a "Value of Statistical Life" (VSL). This methodology has been extensively surveyed (for a recent review see Markandya, 1996). Although there are good reasons for thinking that altemative methods of valuation may be preferable (for example based on the value of life years lost), the VOSL method of valuation has been widely used and has some general acceptance. For the EU countries ExtemE (1995) estimated a central VOSL at ECU 2.6mn ($3.1mn), which is broadly consistent with figures used for the US. This was in 1990 prices. Converting to 1995 prices gives a VOSL of ECU 3.14mn ($3.9mn). PACE (1992) used a VOSL for the US of $4.0mn and Krupnick *et al* (1995) used a value of $3.6mn. For non-OECD countries, such a value is almost certainly too high; it broadly measures individual willingness-to-pay to reduce the risk of death by a small amount.

The above values of VSL can be transferred to other countries through the use of an 'income elasticity' as outlined above. In order to assist researchers in estimating the health benefits of employment, Table 1 provides the VSL for different countries based on an income elasticity of 1 and Table 2 the VSL for an elasticity of 0.35. Both sets of figures use a VOSL for the US of $4.0mn. The PPP GDP per capita for the US is $25,880 based on data from the World Bank Development Report[1].

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[1] *In order to facilitate the comparison of economic activity between countries, the UN's International Comparison Programme (ICP) developed internationally comparable measures of GNP, known as purchasing power parity (PPP) estimates of GNP; these are derived using purchasing power parities as opposed to exchange rates as conversion factors. The PPP conversion factor is defined as the number of units of a country's currency required to buy the same amounts of goods in the domestic market as one dollar would buy in the United States (World Bank, 1996). Data on the average domestic prices of a representative basket of goods and services are collected by the ICP, and PPPs are derived in relation to the average international prices that are implicitly*

4.1 Issues Arising in the Estimation of the Value of a Statistical Life

The main issues that arise with the application of the value of a statistical life in these studies are the following:

a) The validity of the methods used in estimating the value of a statistical life.

b) The distinction between voluntary and involuntary risk;

c) The transfer of risk estimate from different probability ranges.

d) The question of the treatment of acute versus chronic mortality, and more generally the treatment of age dependent mortality.

Validity of different methods of estimating VSL

All three methods of valuing a statistical life have been subject to criticism. The wage-risk method relies on the assumption that there is enough labour mobility to permit individuals to choose their occupations to reflect all their preferences, one of which is the preference for a level of risk. In economies suffering from long-standing structural imbalances in the labour markets this is at best a questionable assumption. Second, it is difficult to distinguish between risks of mortality and morbidity. Third, the WTA will depend on perceived probabilities of death. Almost all studies, however, use a measure of the long-run frequency of death as a measure of risk. This makes the results quoted unsatisfactory. Fourth, the probabilities for which the risks are measured are generally higher than those faced in most of the environmental impacts. This point is returned to below, but a related factor is that the high risk occupations involve individuals whose WTA for an increase in the risk of death is not typical of the population at large (e.g.,

derived from the prices of all participating countries (World Bank, 1996). No data are available for a number of countries that are in politcal turmoil.

steeplejacks)[2]. The net impact of all these factors is difficult to gauge but it is likely that the estimated WTA will be lower than the true WTA.

TABLE 1

Value of Statistical Life for Various Countries

Country	PPP GNP US$ 1994	VSL US $ '000 1995	Country	PPP GNP US$ 1994	VSL US $ '000 1995
ARGENTINA	8.720	1.348	MALAWI	650	100
ARMENIA	2.160	334	MALAYSIA	8.440	1.304
AUSTRALIA	18.120	2.801	MALI	520	80
AZERBAIJAN	1.510	233	MAURITANIA	1.570	243
BANGLADESH	1.330	206	MAURITIUS	12.720	1.966
BELARUS	4.320	668	MEXICO	7.040	1.088
BENIN	1.630	252	MOROCCO	3.470	536
BOLIVIA	2.400	371	MOZAMBIQUE	860	133
BOTSWANA	5.210	805	NAMIBIA	4.320	668
BRAZIL	5.400	835	NEPAL	1.230	190
BULGARIA	4.380	677	NEW ZEALAND	15.870	2.453
BURKINA FASO	800	124	NICARAGUA	1.800	278
BURUNDI	700	108	NIGER	770	119
CAMEROON	1.950	301	NIGERIA	1.190	184
CANADA	19.960	3.085	NORWAY	20.210	3.124
CENTRAL AFR. REP.	1.160	179	OMAN	8.590	1.328
CHAD	720	111	PAKISTAN	2.130	329
CHILE	8.890	1.374	PANAMA	5.730	886
CHINA	2.510	388	PAPUA NEW GUINEA	2.680	414
COLOMBIA	5.330	824	PARAGUAY	3.550	549
CZECH REPUBLIC	8.900	1.376	PERU	3.610	558

(cont...)

[2] This is probably one reason that the estimated value of life declines as the mean risk level in a group increases. From a theoretical perspective one would expect the opposite if the populations were homogeneous.

(continued)

Country	PPP GNP US$ 1994	VSL US $ '000 1995	Country	PPP GNP US$ 1994	VSL US $ '000 1995
DOMINICAN REP	3.760	581	PHILIPPINES	2.740	423
ECUADOR	4.190	648	POLAND	5.480	847
EGYPT	3.720	575	ROMANIA	4.090	632
EL SALVADOR	2.410	372	RUSSIAN FED	4.610	713
ESTONIA	4.510	697	RWANDA	330	51
ETHIOPIA	430	66	SAUDI ARABIA	9.480	1.465
GAMBIA	1.100	170	SENEGAL	1.580	244
GHANA	2.050	317	SIERRA LEONE	700	108
GUATEMALA	3.440	532	SINGAPORE	21.900	3.385
GUINEA-BISSAU	820	127	SLOVENIA	6.230	963
HAITI	930	144	SOUTH AFRICA	5.130	793
HONDURAS	1.940	300	SRI LANKA	3.160	488
HUNGARY	6.080	940	SWITZERLAND	25.150	3.887
INDIA	1.280	198	TAJIKISTAN	970	150
INDONESIA	3.600	556	TANZANIA	820	96
ISRAEL	15.300	2.365	THAILAND	6.970	1.077
JAMAICA	3.400	526	TOGO	1.130	175
JAPAN	21.140	3.267	TRINIDAD & TOBAGO	8.670	1.340
JORDAN	4.100	634	TUNISIA	5.020	778
KAZAKSTAN	2.810	434	TURKEY	4.710	728
KENYA	1.310	202	UGANDA	1.410	218
KOREA	10.330	1.597	UKRAINE	2.620	405
KUWAIT	24.730	3.822	URUGUAY	7.710	1.192
KYRGYZ REPUBLIC	1.730	267	USA	25.880	4.000
LATVIA	3.220	498	UZBEKISTAN	2.370	366
LESOTHO	1.730	267	VENEZUELA	7.770	1.201
LITHUANIA	3.290	509	ZAMBIA	860	133
MADAGASCAR	640	99	ZIMBABWE	2.040	315

Source: World Bank (1996).

Notes: Countries are arranged alphabetically.

Elasticity is assumed to be 1.00

VOSL is assumed to be US $4.0 mn (1995)

TABLE 2

Value of Statistical Life for Various Countries

Country	PPP GNP US$ 1994	VSL US $ '000 1995	Country	PPP GNP US$ 1994	VSL US $ '000 1995
ARGENTINA	8.720	2.733	MALAWI	650	1.102
ARMENIA	2.160	1.677	MALAYSIA	8.440	2.702
AUSTRALIA	18.120	3.531	MALI	520	1.019
AZERBAIJAN	1.510	1.480	MAURITANIA	1.570	1.500
BANGLADESH	1.330	1.415	MAURITIUS	12.720	3.120
BELARUS	4.320	2.138	MEXICO	7.040	2.536
BENIN	1.630	1.520	MOROCCO	3.470	1.980
BOLIVIA	2.400	1.740	MOZAMBIQUE	860	1.215
BOTSWANA	5.210	2.283	NAMIBIA	4.320	2.138
BRAZIL	5.400	2.311	NEPAL	1.230	1.377
BULGARIA	4.380	2.148	NEW ZEALAND	15.870	3.371
BURKINA FASO	800	1.185	NICARAGUA	1.800	1.574
BURUNDI	700	1.131	NIGER	770	1.169
CAMEROON	1.950	1.618	NIGERIA	1.190	1.361
CANADA	19.960	3.652	NORWAY	20.210	3.668
CENTRAL AFR. REP.	1.160	1.349	OMAN	8.590	2.719
CHAD	720	1.142	PAKISTAN	2.130	1.669
CHILE	8.890	2.752	PANAMA	5.730	2.360
CHINA	2.510	1.768	PAPUA NEW GUINEA	2.680	1.809
COLOMBIA	5.330	2.301	PARAGUAY	3.550	1.996
CZECH REPUBLIC	8.900	2.753	PERU	3.610	2.007
DOMINICAN REP.	3.760	2.036	PHILIPPINES	2.740	1.823
ECUADOR	4.190	2.115	POLAND	5.480	2.323
EGYPT	3.720	2.029	ROMANIA	4.090	2.097
EL SALVADOR	2.410	1.743	RUSSIAN FED	4.610	2.187
ESTONIA	4.510	2.170	RWANDA	330	869
ETHIOPIA	430	953	SAUDI ARABIA	9.480	2.815
GAMBIA	1.100	1.324	SENEGAL	1.580	1.503
GHANA	2.050	1.647	SIERRA LEONE	700	1.131
GUATEMALA	3.440	1.974	SINGAPORE	21.900	3.773
GUINEA-BISSAU	820	1.195	SLOVENIA	6.230	2.430
HAITI	930	1.249	SOUTH AFRICA	5.130	2.270
HONDURAS	1.940	1.615	SRI LANKA	3.160	1.916
HUNGARY	6.080	2.409	SWITZERLAND	25.150	3.960
INDIA	1.280	1.397	TAJIKISTAN	970	1.267
INDONESIA	3.600	2.006	TANZANIA	620	1.084
ISRAEL	15.300	3.328	THAILAND	6.970	2.527
JAMAICA	3.400	1.966	TOGO	1.130	1.337
JAPAN	21.140	3.727	TRINIDAD & TOBAGO	8.670	2.728
JORDAN	4.100	2.099	TUNISIA	5.020	2.253
KAZAKSTAN	2.810	1.839	TURKEY	4.710	2.203
KENYA	1.310	1.408	UGANDA	1.410	1.445
KOREA	10.330	2.900	UKRAINE	2.620	1.794
KUWAIT	24.730	3.937	URUGUAY	7.710	2.618
KYRGYZ REPUBLIC	1.730	1.552	USA	25.880	4.000
LATVIA	3.220	1.929	UZBEKISTAN	2.370	1.733
LESOTHO	1.730	1.552	VENEZUELA	7.770	2.625
LITHUANIA	3.290	1.943	ZAMBIA	860	1.215
MADAGASCAR	640	1.096	ZIMBABWE	2.040	1.644

Source: World Bank (1996)
Notes: Countries are arranged alphabetically.
Elasticity is assumed to be 0.35.
VOSL is assumed to be US $4.0 mn (1995).

Voluntary and involuntary risk

There is strong evidence to suggest that individuals treat voluntary risk differently from involuntary risk, with the WTA for a voluntary risk being much lower than that for an involuntary risk. Starr, 1976 has estimated, on a judgmental basis, the difference between the willingness to accept a voluntary increase in risk and an involuntary increase. He finds the latter to be around ten times as high as the former for probabilities of death in the range 10^{-6}-10^{-7}. Interestingly, for lower probabilities that are typical of the impacts of particle pollution, estimates of the differences are not available. In another study of the difference (Litai, 1980), it has been argued that the difference could be as much as 100 times.

The CVM methods are subject to the criticism that the choices are hypothetical and that individuals are not familiar with the concepts of risk involved. Certainly, there have been serious difficulties in conveying the impact of different probability changes through questionnaire methods. Finally, the consumer expenditure approach is subject to the difficulties that perceived probabilities are very different from objective probabilities, and that the effects of the expenditures are to reduce the risk of death as well as of illness following an accident. It is difficult to separate the two impacts in the studies.

Probability ranges for the estimation of VSL

Finally, there is the issue of the probability range over which the estimation is carried out and over which it is applied. Typically one is dealing with much lower probabilities of death in most environmental cases (of the order of 10^{-6} and lower), whereas the studies on which the estimated value of a statistical life is based are dealing with probabilities of between 10^{-1} to 10^{-5}. Furthermore, as the survey by Fisher, Chestnut and Violette (1989) has pointed out, the results from studies at the higher end of the probability range are less reliable. As mentioned earlier, theoretical models would tend to predict that the WTA for lower risks should be lower but, if anything, the empirical literature shows the opposite. Partly this is due to the fact that the groups are not homogeneous. The issue remains unresolved and there is little that can be done about this problem at this stage. In the medium term, research on the theoretical and empirical aspects of the problem is needed.

For all these reasons the studies are likely to be biased, with the wage-risk studies producing values that are too low and the CVM studies values that are too high. Taking an average, as has been done here, is averaging unknown errors. One cannot say what the final impact will be. One can, however, draw some comfort from the fact that the values are, in broad terms, consistent and in a plausible range.

The Treatment of Age Dependent Mortality, Ill Health and Latency Effects

In the case of air pollution from electricity generation the key questions that arise are: (a) should we adjust the VSL values for the fact that many of those affected are old, (b) should some adjustment be made for their state of health and (c) should some adjustment be made for a lapse of time between the exposure and the impact. The analysis of all these issues is relatively recent in the literature and therefore there are not many studies that can be quoted. This section provides, however, a state of the art review of a developing area of research.

Age Dependence for VSL

The issue of age has arisen because some of the studies, and much of the clinical evidence suggests that particle pollution has health effects disproportionately on the elderly. For example, Schwartz and Dockery (1992) report a relative risk for under 65s as 1.049 per 100$\mu g/M^3$ of PM_{10} and for over 65s as 1.166. Other studies that look at age as a distinct variable also find this effect.

The literature on age and VSL points to a relationship that is non-linear. The VSL increases with age in the early years and then declines, with a peak value at 40-50 years of age. This is supported by both theoretical and empirical studies. In turn, several empirical studies have produced evidence of a significant inverse relationship between the VOSL and age, at least beyond middle years, perhaps the most marked example being the pronounced inverted-U life-cycle for the roads VOSL which emerged from the data generated by a nationally representative sample survey employing the contingent valuation (CV) approach carried out in 1982 and reported in Jones-Lee (1989). The results from that study are summarised in Table 3 below.

TABLE 3

Estimates of VSL for different ages as a percentage of VSL at age 40

Age	20	25	30	35	40	45	50	55	60	65	70	75
VSL as % at age 40	68	79	88	95	100	103	104	102	99	94	86	77

Source: Jones-Lee et al (1985).

All these issues about the relationship between age and VSL lead inescapably to the conclusion that VSL **should be adjusted for age.** The above table could be used to adjust VSL for age, **if data on health impacts were available on an age basis.** Unfortunately, this is rarely the case in the empirical literature.

Impact of Health Impairment

Apart from the effects of age, one might expect VSL to vary with the state of health. There are two dimensions to this. One is the effect of pure health impairment and the other is the effect of shortening of life span. If a person's quality of life is poor this may effect his or her WTP for a reduction in the risk of death. There is little evidence, however, that points to this, although health service professionals do use a 'Quality Adjusted Life Years' (or QUALY) approach in which resources are allocated on the basis of paying no more than a certain amount for a QUALY. We retum to this method below. At this point we simply note that the VSL approach does not adjust for pure health impairment. Nor does it adjust for reductions in life expectancy. For particle pollution this is particularly important, because there is a lot of clinical experience to suggest that the life expectancy of those who die from such exposure is already very short, perhaps only a few months.

Most observers agree that it is inappropriate to take a value for VSL based on a population with normal life expectancy and apply it to a population with a much shortened life expectancy. One way to approach this is to value life years directly. The issue is of particular importance when the impacts of air pollution are classified as "acute mortality". For such cases the mechanism is the number of air pollution days contributing to a higher number of deaths on the same day or on immediately following days. In this case, the 'at-risk' population consists mainly of elderly people (>65 years of age) with existing (serious) cardio-respiratory problems. The expectation is that persons affected are already quite ill and have only a short life expectancy.

The other kind of mortality impact is classified as chronic mortality. Here the mechanism here is long-term exposure to air pollution, which leads to disease, which in tum contributes to premature death. In this case, it is formally irrelevant whether death follows a higher pollution day. Cohort studies generally show increased mortality from cardio-respiratory disease, and from lung cancer.

The acute effects of various pollutants across a range of health endpoints are reasonably well established. These include respiratory infections, asthma attacks and restrictive activity days. Research has tried to establish reliable exposure-response functions for such effects. It is more difficult to establish relationships for chronic effects such as bronchitis or other longer term respiratory infections.

Impacts of Latency on VSL

If exposure to particle pollution today causes the risk of death to increase T years from now, the WTP to avoid that risk is not the same as that associated with an increase in the risk of death now. The accepted way to deal with such latency is to discount future risks, so that if the WTP for an immediate reduction in risk is \$X, then the WTP for a reduction in a risk with a latency of T years is $\$X.(1+r)^{-T}$. The key question, of course, is what value should r take?

In ExternE (1995) this issue has been discussed at great length. It is noted there that there is a case for relatively high rates (around 11%), as well as one for low rates, in the region of 3%. Given the lack of agreement among economists as to which rate is the appropriate, rate it is recommended that calculations be done with both rates and the resulting range of values reported.

Value of Life Years Lost

An alternative approach to analysing changes in the risk of death is to look at them in terms of the WTP for life years and to report a value for a life year lost (VLYL). The advantage of such a method is that it allows greater flexibility in valuation, and, furthermore, one that clinicians are more comfortable in estimating. It also brings the WTP approach closer to the QUALY (quality of life years) approach which is widely used in health planning work. This has been studied and developed by Moore and Viscusi (1988), and Johannesson and Johansson (1996)

If we are to use VLYL what numbers should we use? As a first approximation it is reasonable to take a constant VLYL over the remaining lifetime of the person and to assume that the VSL for that person is the expected discounted present value sum of future life years. Thus, if VSL is \$4 million for a person of 40 years of age, and if the probability of a person of age 40 surviving to age t is $p_{40,t}$, then the VLYL is the solution to equal to:

$$VSL = VLYL \sum_{t=40}^{t=100} (1+r)^{-t}\, p_{40,t}$$

Such a calculation has been carried out below based on survival probabilities for European males and are shown in Table 4. The values emerging are around $200,000 with a discount rate of 3% and around $415,000 for a discount rate of 11%.

TABLE 4

VLYL Values for European males in good health

Discount Rate	VLYL with $4mn VSL for age 35	VLYL with $4mn VSL for age 45
0%	$107,134	$147,770
3%	$179,746	$214,648
11%	$402,620	$430,710

The transfer of these values to industrialising countries such as Brazil can be based on the same coefficients as were applied to VSL. Thus Tables 3 and 4 can be used to scale VLYL figures in the same way. It is desirable for the VLYL values to be calculated for the survival probabilities appropriate to the country concemed. If these are very different from the ones in Europe, the VLYL values will differ for that reason as well.

These values can be used for both acute and chronic mortality impacts. In each case an estimate has to be made of the number of life years lost per unit exposure. This information has to be provided by the epidemiologists.

5 The Valuation of Morbidity Effects

Epidemiological data has identified a relationship between certain health 'endpoints' and particle pollution. The following are the endpoints for which some valuation is, therefore sought:

- Bronchodilator use in asthmatics
- Cough in asthmatics
- Lower respiratory symptoms in asthmatics (wheeze)
- Prevalence of child bronchitis
- Prevalence of child chronic cough
- Restricted activity days
- Chronic bronchitis in adults
- Hospital admissions for congestive heart failure
- Chronic admissions for ischaemic heart disease
- Respiratory hospital admissions
- Cerebrovascular hospital admissions

The full cost for an illness is composed of the following parts: (a) the value of the time lost because of the illness,(b) the value of the lost utility because of the pain and suffering, and (c) the costs of any expenditures on averting and/or mitigating the effects of the illness. The last category includes both expenditures on prophylactics, as well as on the treatment of the illness once it has occurred. To value these components researchers have estimated the costs of illness, and used CVM methods as well as models of avertive behaviour.

The costs of illness (COI) are the easiest to measure, based either on the actual expenditures associated with different illnesses, or on the expected frequency of the use of different services for different illnesses. Part of these costs may be incurred by the individual directly and others through private insurance or through general taxation. In the many countries, a significant portion of the costs of respiratory illness (at least of the serious kind) are paid for through general taxation. The use of COI measures in estimating the costs of air pollution has been carried out in Brazil by Seroa da Motta and Fernandes Mendes (1996). These figures are useful for policy purposes but, as the discussion in this section shows, the full morbidity costs should be taken to be higher than the COI costs.

The costs of lost time are typically valued at the post-tax wage rate (for the work time lost), and at the opportunity cost of leisure (for the leisure time lost). Typically the latter is between one half and one third of the post-tax wage. Complications arise when the worker can work but is not performing at his full capacity. In that case an estimate of the productivity loss has to be made.

It is important to note that COI is only a component of the total cost and, furthermore, it is not necessarily a part of the WTP to avoid an illness. For example, if a person's medical costs are paid for through general taxation, the stated WTP to avoid a particular health 'endpoint' will not include such costs. Hence the relationship between COI and WTP are complex, and one cannot add the two items together to arrive at the total cost. In part this relationship has been studied, by making a direct comparison of the two estimates and looking at their ratio. Rowe et al. (1995) have done this for US data and find that the ratio of WTP to COI is in the range 1.3 to 2.4. On the basis of their analysis they recommend a value of 2 for adverse health effects other than cancer and a value of 1.5 for non-fatal cancers. To arrive at the total cost of an illness, however, one should take WTP, **plus** the part of COI that is not reflected in WTP. This will be the component that is paid for through taxation and, possibly, through insurance. Even in the US, some 68% of health costs are paid for by third parties.

Although the relationship between COI and WTP is complex, it offers one method of arriving at a realistic cost figure for morbidity endpoints, for many of which we do not have any WTP studies. In this section we report on morbidity estimates from three sets of figures: (a) US and European WTP studies for certain endpoints, (b) UK COI estimates 'grossed up' for the difference between WTP and COI.

The WTP for health endpoints can be measured either through the CVM approach, or through models of avertive behaviour. The latter involves the estimation of a 'health production function', from which one estimates the inputs used by the individual in different health states, and taking the difference in value between these obtains the cost of moving from one health state to another. The difficulty is in estimating that function, where many 'inputs' provide more than one service (e.g., bottled water, air conditioners), and where the changes in consumption as a function of the state of illness are difficult to estimate. There are few estimates of health endpoints based on such models, and none for the industrialising countries.

In the ExternE work, we took values of endpoints from US studies, adjusted them for inflation and converted into ECU. For other endpoints, where data were available from European studies. In Table 5 we report these figures. The figures reported here are from ExternE (1995), updated from some recent US work. It would be interesting to compare the costs implied by these figures with those obtained by, for example, Seroa da Motta, and Fernandes Mendes (1996).

In making these and other transfers, they key issue is their validity in local conditions. Transferability is most questionable when medical service costs influence the WTP. Since the provision of health services is different in different countries, and costs vary a lot, simply converting the US costs into Korean or any other costs at the PPP adjustment factors as given in Tables 3 and 4 must be considered as a doubtful practice. For this reason the above transfer is recommended only for those US studies for which the WTP is not medical cost sensitive (restricted activity days, cough days, symptom days and prevalence of chronic cough). For endpoints involving hospital treatment we have suggest taking local health cost data and scaling up on the basis of the ratio of COI to WTP.

Where an estimate of the value can be made on the basis of PPPGDP the correction factors have to use the EU PPPGDP as the baseline value and not that of the US. Thus the estimate of values to country I are:

$WTP_{EU}.(PPPGDP_I/PPPGDP_{EU})^{0.35}$ for the case of an income elasticity of 0.35, and

$WTP_{EU}.(PPPGDP_I/PPPGDP_{EU})$ for the case of an income elasticity of 1.

The $PPPGDP_{EU}$ is $17,900. Thus, for example, a case of a cough in asthamatics is valued in Korea as $\{8.(10330/17900)^{0.35}\}$ for an income elasticity of 0.35 and $\{8.(10330/17900)^{1}\}$ for an income elasticity of 1.0. This gives the values of $6.6 and $4.6 respectively.

6 Conclusions on the valuation of health impacts

6.1 Summary of results

This paper has reviewed the different studies of the costs of mortality and morbidity arising from air pollution and has made some broad recommendations of values.

For mortality the 'Value of a Statistical Life' (VSL) has been taken at $4 million from the risk literature, with high bounds of uncertainty (perhaps a factor of 10). For industrialising countries such as Brazil, Tables 3 and 4 provide approximate values, based on a real income adjustment.

An altemative to the VSL is the VLYL, or value of a life year lost. Based on the VSL of 2mn, the VSL would (depending on the discount rate) amount to between $160,000 and $330,000. Transfer of these to industrialising countries can be made on the same basis as for VSL – i.e. using Tables 3 and 4 and replacing the $4 mn with the appropriate value of VLYL.

TABLE 5

Morbidity health endpoints and their valuation (Values are in $1996)

Endpoint	(ExtemE and this study)	Comments on transferability
Bronchodilator use in asthmatics	40	Use the PPPGDP adjustment factors given in the main text.
Cough in asthmatics	8	
Lower respiratory symptoms in asthmatics (wheeze)	8	
Prevalence of child bronchitis	237	
Prevalence of child chronic cough	237	
Restricted activity days	80	
Chronic bronchitis in adults	256,000	
Hospital admissions for congestive heart failure	5,760	These values are of the UK, based on UK health costs, grossed up for the ratio of WTP to COI. A similar exercise can be carried out in each country, based on COI data.
Chronic admissions for ischaemic heart disease	5,760	
Respiratory hospital admissions	3,520	
Cerebrovascular hospital admissions	5,760	

Source: Externe (1995), updated for present study.

There is no agreed discount rate for latent effects and for effects that will be spread out over a number of years. There is a case of a rate of around 1-3% and for a rate of around 8-11%. There is also a case for a time variant discount rate, but this needs further research for the precise parameters to be established.

For morbidity impacts the key endpoints arising from air pollution have been identified. The estimation of the money values of these is based on a number of sources. For non-hospital related endpoints, studies from the US and one from Norway have been considered and values derived. For hospital related endpoints UK Department of Health figures have been adjusted for the difference between the 'Costs of Illness' (COI) and the WTP. In the case of an industrialising country transfers of health endpoints that do not involve extensive medical costs can be made by using the PPPGDP adjustment, as indicated in the text. For other health points we recommend the direct estimation of the costs of the illness and then a scaling up by the ratio of WTP to COI.

6.2 How should these results be used

The purpose of this approach is to aid decision-makers in making better decisions – i.e. ones that use scarce resources more efficiently. While such valuations of resources in areas other than health raises few objections when it comes to health issues, especially mortality, the critics are more vociferous. This was made clear, for example, in the last IPCC report (IPCC, 1996), where the valuation of damages required an estimate of loss of life from the impacts of global warming. The authors responsible for that section of the report took values somewhat similar to those suggested in Table 1 -- i.e. based on an income elasticity of one. National govemments, however, objected to this, arguing that it was not appropriate to value the deaths of the citizens of a poorer country differently from those of another (richer) country. In other words they rejected the application of the above methodology for the purposes of valuing climate change effects.

This example serves to make an important point. The purpose of valuing health impacts, as was noted at the beginning of this paper, is to provide a better basis for decision-making. In the case of climate change, no one would wish to arrive at a policy in which the critical factor was the different values attached to the deaths of the rich and the poor. That would, rightly, be seen as immoral. Hence the appropriate thing to have done would have been to value all deaths at the world GDP average. Indeed, when the VSL method is applied at the national level – e.g. for transport planning – no one even thinks of taking different values for the deaths of the rich and the poor citizens. A single national value is applied. It is important, however, that the correct national value be applied. If it too high, we will devote too much to safety and if it is too low we will not devote enough.

The values provided in this paper are guides to the valuation of health impacts in national programmes of investment. They are not in tended to serve as comparisons between nations. Nor are they to be used in policies involving global pollutants. A good example of how they might be used, is to compare the costs of programmes of health improvement with the lives saved and obtain the 'cost per life saved'. This has been done for Brazil recently by Seroa da Motta (1995). Looking at the costs of drinking water treatment, sewage collection and sewage treatment, he finds a cost per life saved in Brazil of around $18,000. From the figures obtained as an average for Brazil in Tables 3 and 4 this is very small; the estimate on the basis of an elasticity of one for the value of a statistical life is $835,000. Hence such programmes of water quality improvement are amply justified. Of course, if funds are not available to finance all projects in which the value of a life saved is less than $835,000, then the govemment has to chose between them. In that event, the programmes with the lowest cost per life saved should be implemented first.

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