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Technical Report on Methodology: Cost Benefit Analysis and Policy Responses

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Abstract

The economic assessment of priorities for a European environmental policy plan focuses on twelve identified Prominent European Environmental Problems such as climate change, chemical risks and biodiversity. The study, commissioned by the European Commission (DG Environment) to a European consortium led by RIVM, provides a basis for priority setting for European environmental policy planning in support of the sixth Environmental Action Programme as follow-up of the current fifth Environmental Action Plan called ‘Towards Sustainability’. The analysis is based on an examination of the cost of avoided damage, environmental expenditures, risk assessment, public opinion, social incidence and sustainability. The study incorporates information on targets, scenario results, and policy options and measures including their costs and benefits.

Main findings of the study are the following. Current trends show that if all existing policies are fully implemented and enforced, the European Union will be successful in reducing pressures on the environment. However, damage to human health and ecosystems can be substantially reduced with accelerated policies. The implementation costs of these additional policies will not exceed the environmental benefits and the impact on the economy is manageable. This requires future policies to focus on least-cost solutions and follow an integrated approach. Nevertheless, these policies will not be adequate for achieving all policy objectives. Remaining major problems are the excess load of nitrogen in the ecosystem, exceedance of air quality guidelines (especially particulate matter), noise nuisance and biodiversity loss.

This report is one of a series supporting the main report: *European Environmental Priorities: an Integrated Economic and Environmental Assessment*. The areas discussed in the main report are fully documented in the various *Technical reports*. A background report is presented for each environmental issue giving an outline of the problem and its relationship to economic sectors and other issues; the benefits and the cost-benefit analysis; and the policy responses. Additional reports outline the benefits methodology, the EU enlargement issue and the macro-economic consequences of the scenarios.

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Reports in this series have been subject to limited peer review.

Prepared by D.W. Pearce, A. Howarth (EFTEC)

The following sections are the supporting documents to the benefit assessment and policy assessment papers in the main report. *Section 1* describes the benefit assessment procedure applied to the eleven environmental issues in this study. *Section 2* concentrates on the nature of economic instruments and the criteria used to select economic instruments. This is followed by a brief typology of economic instruments and finally this section makes the first step towards matching policies to the environmental issues considered in this study. *Section 3* introduces monetary valuation of ‘non-marketed’ environmental goods, such as clean air, clean water, etc. The concept of ‘total economic value’ is discussed, followed by a brief description of the valuation techniques used. This study relies heavily on the process of ‘benefits transfer’ (BT) which involves taking existing monetary valuation studies (i.e. ‘willingness-to-pay’ values) and applying them outside the site context where the study was originally conducted. *Section 4* describes the adjustments involved in the benefits transfer process and lists the main criteria for successful and accurate benefits transfer. *Section 5* presents the analytics and the empirical evidence of the income elasticity of demand for the environment. This information is used to calculate benefits in the future (i.e. in 2010), it is assumed that environmental quality has a rising relative price through time that is linked to growth in income per capita. *Section 6* introduces the importance of risk valuation in environmental cost-benefit analysis. The concept of a ‘value of statistical life’ (VOSL) is discussed, a brief discussion of the techniques used to estimate VOSL is given followed by empirical evidence of the VOSL. This section also discusses the ‘value of life year’ (VOLY) as well as looking at others’ valuation of risk to individuals, the value of future lives and the affect of unequal income distribution on the VOSL.

Contents

1. BENEFIT ASSESSMENT PROCEDURE

General methodology

The general structure of the benefit assessments for the different environmental problems is as follows:

Importance of the issue
A brief discussion stating how public and expert opinion rank the issue as a serious environmental problem is provided.

Monetary valuation
The measurement of benefits is essentially the measurement of avoided damages. Since the scenarios (other than Accession) generally simulate overall improvements in the environment, benefits will tend to get larger as we move from the baseline to the AP or TD scenarios. Where D refers to environmental damage, we estimate the benefits for the AP and TD scenarios in the following way:

$$D_{BASE} - D_{AP} \quad = \quad \text{Benefits of AP}$$

$$D_{BASE} - D_{TD} \quad = \quad \text{Benefits of TD}$$

Figure 1.1 gives a stylised illustration of the benefit in 2010 of the AP scenario over baseline.

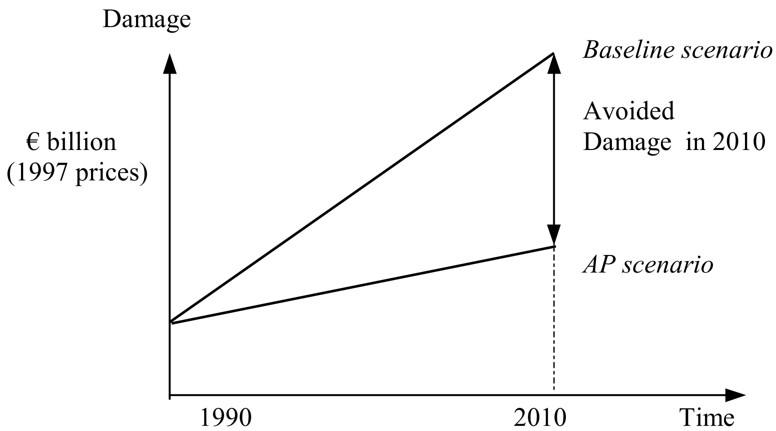

Figure 1.1: Benefit of AP scenario in 2010

Benefits are defined in terms of individuals' willingness to pay (WTP) to secure the benefits. Ideally a wider concept of benefits would include the macroeconomic benefits. Essentially, the difference between the WTP and macroeconomic benefits is one of scope. WTP tends to capture 'partial' benefits. These will approximate total benefits if the policy measures make only marginal changes to the economy. However, in terms of the scenarios adopted, the policy measures are not marginal, but involve fairly significant discrete changes. Thus, the wider concept of benefit is embraced allowing for the feedback effects of the policy on other prices and quantities in the economy.
This is possible only for those measures relating to the energy sector where the GEM-E3 model is used. In other cases, no macroeconomic model exists or the measures in question cannot easily be incorporated in such models (e.g. coastal waters, biodiversity etc).

Relevant WTP values for each environmental problems are drawn from an extensive literature review of the most recent and relevant monetary valuation studies conducted for this study. These give WTP estimates for environmental improvement or WTP estimates to avoid environmental damage. Two main groups of monetary valuation techniques are used, stated and revealed preference techniques (for further details refer to Section 3 on monetary valuation techniques). Taking existing monetary valuation (WTP) studies and applying them outside of the site contexts where the study was originally carried out requires extensive use of 'benefits transfer' (see Section 4 on benefits transfer).

In order to calculate benefits in the future (i.e. in 2010), we assume that environmental quality has a rising relative price through time that is linked to growth in income per capita. The following formula is adopted to adjust the valuations accordingly:

$$WTP_{2010} = WTP_{1990} \cdot (Y_{2010}/Y_{1990})^{e}$$

Where *WTP* refers to willingness to pay valuations, *Y* is EU per capita GNP (assumed as: Y_{2010} = € 19830 and Y_{1990} = € 14247, source RIVM) and *e* is the income elasticity of demand (assumed here as 0.3 refer to Section 5 on income elasticity of demand). An annual increase in relative prices for environmental quality of 0.5% per annum is arrived at[1].

The likely time paths of benefits (and costs) are known in only a few cases, thus the benefit results are reported for a representative future year only, i.e. 2010. It is acknowledged that different time paths will produce potentially different results. When benefits are later compared with costs, the net benefits clearly depend on <u>what</u> is done, the <u>scale</u> of control measures and on the <u>time paths</u> of these measures. However, the research team is of the view that no major divergence of results will occur because of the choice of a representative year for benefits and costs.

Benefit estimates are summarised in Table 1.1 for those environmental problems with clearly defined AP / TD scenarios, such as climate change, acidification, tropospheric ozone, waste management, Human health, air quality and noise and nuclear risks. The values relate to benefits to the EU15 only unless otherwise stated. All values are benefits in 2010 only and are given in terms of € (1997 prices). For a more detailed discussion of the benefit estimates, refer to the benefits assessment for each environmental problem given in the Technical Reports.

[1] Ideally, it should be decomposed by country since income per capita varies by country. However, we suggest EU-wide approximations are suitable.

Table 1.1 Summary of benefit estimates for each environmental problem

Environmental problem	Primary benefit € billion	Secondary benefit € billion
Climate change		
NT-AP	3.7*	20.5 (11.5)
FT-AP	3.7*	13.4 (7.5)
Acidification		
NT-AP	21.7 (14.0)	7.1 (1.6)
FT-AP	25.2 (16.3)	7.8 (1.7)
TD	58.9 (38.1)	12.6 (1.7)
Tropospheric ozone		
NT-AP	5.6 (0.7)	-
FT-AP	5.7 (0.7)	-
TD	9.1 (1.2)	-
Waste management		
AP with source reduction	8.7	0.4
AP without source reduction	7.2	0.4
TD max compost and recycle	10.3	-
TD max incineration	-2.8	-
Particulate matter		
AP	5.3 (3.1)	-
TD	(24.2 (14.0))	-
Nuclear risks		
TD	6.8	-

Where: '-' = not available, NT = No Trade, FT = Full Trade, AP = Accelerated Policy scenario, TD = technology driven scenario. * benefit to world. Estimates given in brackets assume premature mortality is valued with VOLY, estimates not bracketed assume VOSL.

Note, primary benefits relate to the control of the pollutants, P_X, causing the environmental problem *X*. Primary benefits for climate change are due to the control of CO_2, CH_4 and N_2O only. For acidification, primary benefits are due to control of SO_x, NO_x and NH_3, for tropospheric ozone primary benefit estimates are due to the direct control of VOCs only and for Human health, air quality and noise, primary benefits relate to the end-of-pipe measures that reduce concentrations of primary PM_{10}.

However, the control of P_X pollutants can lead to the control of other pollutants *Py* causing other environmental problems Y. This effect is known as the secondary benefit of measures to control X.
The secondary benefits of climate change control are to acidification, low level ozone and Human health, air quality and noise (through the control of primary PM_{10} and secondary aerosols). Note that the secondary benefits due to reductions of secondary aerosols are not estimated separately, they are already accounted for in the secondary benefits to acidification. The secondary benefits of acidification control are to tropospheric ozone and Human health, air quality and noise (i.e. through the control of primary PM_{10} and secondary aerosols). Note that the secondary benefits due to reductions in secondary aerosols are subsumed in the primary benefit estimtes for acidification. The secondary benefits of waste management are to climate change control.

Uncertainty

Uncertainty is endemic to this study. The main sources of uncertainty in the benefit assessment approach are:

- scientific uncertainty about the impacts of given pressures on the state of the environment;
- economic uncertainty about the willingness to pay of the relevant population to avoid the impact;

Since these uncertainties are unavoidable the relevant approach is one which tries to estimate central tendencies and the confidence interval around that central tendency. Even though uncertainties at the various stages of the analysis may be multiplicative, estimates of central tendency will tend to remain unchanged, although the dispersion about the mean will increase.

It is tempting to think that avoiding some of the stages of the analysis may reduce uncertainty. For example, monetary valuation (WTP) of premature mortality may be subject to confidence ranges which increases the range of uncertainty attached to the impact measure (e.g. lives lost). Casual commentators suggest that the monetary valuation stage should therefore be avoided. This is a mistaken strategy. First, the addition of the monetary valuation stage does not change the mean outcome. Second, the wider confidence interval that may emerge does indeed 'increase' the uncertainty of the estimate of effect, but if the monetary valuation stage is avoided other forms of uncertainty are added to the picture. Pursuing the pollution-health example, the analysis may be presented in terms of costs and lives prematurely lost, or it may be presented in terms of costs and the monetised (economic) value of lives prematurely lost. The former avoids the monetary valuation estimate; the latter explicitly includes it. But while the latter adds to uncertainty in the sense of increasing the confidence interval, the former increases other forms of uncertainty. Using the former, i.e. 'lives lost prematurely' assumes, for example, that all lives are to be equally weighted regardless of the length of life expectancy lost, or the health state of those at risk etc. If this is not what is desired, then lives lost prematurely can be weighted by life expectancy and health state. But in so doing, the analyst superimposes a weighing on the indicator that has nothing to do with the perceptions or preferences of those at risk. In short, a new form of uncertainty is introduced, namely the uncertainty about the extent to which indicators are now responsive to individuals' wants and desires, the basic value axiom of welfare economics.

Avoiding the monetary valuation stage also creates other forms of uncertainty. Where the impacts are measured in non-monetary units and compared to costs, there is no guideline on whether a policy is worth undertaking. Monetisation provides the guideline that policies should at least past a test to the effect that benefits should exceed costs. Cost-effectiveness indicators have no such test since it is never possible to tell whether an incremental unit of effectiveness is worth the cost of securing it. (Note also, that selecting any target of effectiveness, e.g. an upper limit on cost per life saved, automatically implies a monetary benefit estimate).

Avoiding the monetary valuation stage may seem like a rational response to the uncertainty embedded in the benefit estimates, but such a response adds at least two other forms of uncertainty. Firstly, the 'democratic' uncertainty, this is the extent to which any outcome is now responsive to individuals' preferences, and secondly the 'decision-making' uncertainty, i.e. the extent to which rational trade-offs between costs and benefits can be made.

The reliability of the WTP values used in this study is tested, where possible, by using confidence intervals around the mean value. Table 1.1 reports mid values only, for ranges refer to the benefit estimates for each environmental problem given in the Technical Reports.

The monetary valuation of premature mortality is a key area of uncertainty in this analysis. Where the probability of death is the same for all age groups in the population we adopt the value of a statistical life relevant to the general population, i.e. € 3.31 million (from € 2.6m 1990 prices converted to 1997 prices using the deflator 1.274). In those areas where deaths are mainly confined to the older age groups in the population we use a reduced VOSL, i.e. 70% of € 3.31 million = € 2.32 million.

For some environmental problems, fatalities occur over a long period, i.e. 1990-2010-2050 (i.e. nuclear risks) and thus the VOSL relevant to 1990 will not be relevant to the whole period. Rather, we would expect VOSL to rise as incomes rise. The effect of income growth is captured by introducing a rising relative price of risk aversion of 0.5% per annum, although there is limited information on the increase over time in the relative 'price' of risk. For further information regarding the issue of premature mortality valuation see Section 6 on valuing statistical live.

Sensitivity
A number of assumptions are made for each separate benefit assessment. Some may have a significant effect on the results, while others will make only a minor difference. For purposes of transparency the key assumptions are stated clearly throughout. In order to see the effect on the net results if these assumptions are changed we conduct a sensitivity analysis. Thus, changes in the key assumptions and the associated quantitative effects are also reported.

2. POLICIES AND ENVIRONMENTAL PROBLEMS

Each environmental problem considered in this study contains a set of quantitative or qualitative 'targets' and each target corresponds to a scenario. Targets might be set in terms of given reductions in emissions, areas of land conserved for biodiversity etc. In order to inform the design of environmental policy, we need some idea of what policy instruments are best suited to the achievement of the targets.

The nature of policy instruments

A critical goal of policy towards the environment is cost-effectiveness: the achievement of the policy goal at least cost. In welfare economics terms, 'least cost' means least loss of economic wellbeing[2]. A narrower goal would be to measure costs solely in terms of the costs borne by the regulated agent in complying with the policy.

There are general reasons for supposing that *economic instruments* are best suited to achieving the least cost goal. Definitions of economic instruments (EIs) are not easy to provide. All forms of regulation impose a cost on the regulated agent, so that the presence of a financial incentive is not peculiar to economic instruments. It is widely argued that EIs leave the regulated agent with more flexibility on *how* to respond to policy. Thus, traditional 'command and control' (CAC) regulation might be regarded as setting target (what to achieve) and mechanism (how to achieve it), whereas EIs leave the regulated agent with the choice of what to achieve and how to achieve it, provided the overall policy goal is met in the aggregate. Thus, an individual regulated agent can emit pollution up to any level provided it pays the necessary environmental tax or holds the necessary permit to emit. The choice of the mix of abatement measures and tax payments/permit holdings is up to the regulated agent. But policy will have set an aggregate goal, for example a total level of emissions, that must be met and permits will be issued equal to this aggregate goal, or an estimate will have been made of the emission reduction effect of taxes so as to achieve the goal.

There are general reasons for supposing that EIs are best suited to achieving the least cost goal. However, the presumption that EIs are more cost effective than CAC is not always the case. In general, quite specific conditions have to be present for EIs to perform better than CAC[3]. These factors need to be taken into account in deciding the 'match' of policy instruments to environmental problems.

Criteria for selecting policy instruments

Fundamental to this study is the use of 'welfare economics', it is therefore appropriate that the criteria for selecting 'desirable' policy instruments should be based on social cost benefit analysis. However, it is important to assess policy instruments against other considerations, such as distributional concerns (i.e. impacts to socio-economic class and region), macroeconomic issues (competition and employment effects), administrative feasibility[4] and subsidiarity (i.e. the 'optimal jurisdiction' issue, in other words, where policy is most effectively located). Subjecting policy instruments to many criteria for acceptability risks making almost all policy instruments fail. Similarly, we have no clear criteria (meta-criteria) for deciding which criteria are the most important. In order to identify rational policy instruments to meet AP scenario targets, we suggest that there are five groups of criteria for choosing a policy instrument, these are set out below:

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[2] Which, ideally, would be measured by the change in the sum of producers' and consumers' surpluses. In practice, this measure will be available in only some cases.

[3] See C.Russell, P.Powell and W Vaughan, Rethinking advice on environmental policy instrument choice in developing countries, Paper to World Congress on Environmental Economics, Venice, June 1998.

[4] Note that, 'political feasibility' is not explicitly considered, since the research team's concern is to define a potential menu of policies. The extent to which such policies are politically feasible is not for the research team to judge.

- causal
- efficiency
- equity
- macro-economic
- jurisdictional

The causal criterion answers the basic question: 'does the policy instrument address the underlying economic failure'? If policy does not address the real causes of environmental degradation, it will have a high risk of failure. It is important to note that real causes do not equate with 'pressures' in the DPSR paradigm, nor what is popularly understood by 'driving forces'. The underlying causes are i) market failures (i.e. not well defined property rights, missing markets and lack of information; ii) intervention failure (i.e. counter-effective subsidies and inconsistent policies; iii) implementation failures, i.e. if legislation exists, but is not fully implemented by Member States, iv) growth of real income, and v) population change, i.e. natural growth, migration and social change. Overall policy measures are targeted at the first three underlying causes only.

The economic efficiency criterion includes: i) benefit cost ratios, ii) cost-effectiveness, iii) benefits, and iv) public opinion for each policy instrument. The least-cost action is embodied in the cost-benefit approach and in cost effectiveness. Public opinion is included in efficiency because public opinion indicates public preferences, which in turn underlie the notion of willingness to pay. WTP is the building block of the benefits assessment.

The equity or distributional criterion considers: i) intra-generational equity (impacts to current socio-economic class, economic sector and region) and ii) inter-generational equity (distributional impacts between generations).

The macro-economic criterion is mainly centred on the NTUA modelling of climate change policy. Policy instrument impacts considered are EU employment, GNP loss and competition effects. The macro-economic impacts are in the final report, see *Chapter 4 Section 4.1*, and for further details refer to *Technical report on Socio-economic trends, macro-economic impacts and cost interface*.

The jurisdictional criterion concentrates on the issue of subsidiarity, i.e. where is policy most effectively located, such as, EU, national or local level. There are three main criteria upon which the level of subsidiarity can be assessed: i) gains from co-operation, ii) gains from harmonisation and co-operation and iii) gains in sustainable implementation.

Types of instruments

The list of instruments is potentially very large. Here we categorise them according to OECD classifications and the discussion in Panayotou (1998)[5].

Command and control

- ambient based standards (e.g. μg of pollutant per m^3);
- emission based standards (e.g. g of pollutant per km travelled in test conditions);
- product based standards;
- technology based standards (BAT, BATNEEC), and
- bans.

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5 See OECD, *Managing the Environment: the Role of Economic Instruments*, OECD, Paris, 1994, and T.Panayotou, *Economic Instruments for Environmental Management and Sustainable Development*, Earthscan, 1998, forthcoming.

Standards may be based on a per pollutant basis or on Integrated Pollution Control (IPC) considerations such that impacts on different environmental media are considered.

Economic instruments

Property rights
Property rights should be secure (enforceable) and transferable for economic efficiency to be assured. They will need to be attenuated in some form (ie certain uses will be forbidden) if they give rise to excessive externalities. Property rights can be private, communal or public, with a presumption that private and communal rights are to be preferred.

Fiscal instruments
- emission taxes (e.g. SO_x charge);
- effluent taxes (e.g. wastewater charge);
- input taxes (e.g. pesticides or fertiliser tax);
- (final) product taxes (e.g. packaging tax);
- export taxes/import taxes;
- differential taxation (e.g. leaded/unleaded gasoline);
- royalty (rent) taxation (e.g. forest taxation);
- land use taxes (taxes vary according to land use);
- accelerated depreciation (environmentally beneficial investments allowed to depreciate faster for tax offset purposes);
- subsidy removal (where subsidies harm the environment, e.g. CAP reform), and
- subsidies (where subsidies benefit the environment) (e.g. subsidies to renewable energy).

Pollution taxes are formally equivalent to pollution charges so that no distinction between the two is made here. But charges and taxes otherwise differ: charges are for the use of a service, whereas taxes tend to raise revenue. The equivalence of charges and taxes in the pollution case arises because the polluter is using a public service - the assimilative capacity of the environment. In administrative terms the more important distinction is that taxes always form part of the fiscal structure and have therefore to be agreed by and administered by the tax authorities. Charges can be outside the control of tax authorities (inland revenue and customs).

Environmental charges
- user charges (e.g. entry fees to protected areas, road pricing);
- betterment charges (charges on properties which benefit from public infrastructure or environmental improvement), and
- impact charges (the obverse of betterment, ie charges on properties for making the environment worse, usually levied when property or land use changes).

Deposit - refund schemes and performance bonds
Here the charge is made in advance of any damage occurring, and refunds are given when the product is safely disposed of or recycled or the environmental degradation is made good. Bonds act in the same way: the bond has to be purchased at the onset on economic activity (e.g. quarrying) and can only be redeemed when there is an indication that restoration has occurred.

- deposit-refund (tax-subsidy) schemes (e.g. on returnable bottles and cans);
- environmental performance bond (e.g. mining, quarrying, forest logging, waste arisings), and
- accident bonds (e.g. for oil spills).

Liability systems
Liability systems rest on the threat of legal action in the event of non-compliance. The charge is collected only in the event of damage occurring and liability systems thus have similarities with

bonds (above). However, bonds collect the charge early on and refund it later. Liability systems collect the charge only in the event of damage.

- legal liability (is ‘strict’ when libaility exists regardless of precautions taken, and is ‘negligence’ when actions taken to avoid damage are taken into account);
- non-compliance fines (charges at some penal rate for emissions above standards);
- joint and several liability (any one contributor to damage can be held responsible for all damage), and
- liability insurance (insurance market premia in the event of damage in a liability context).

Financial incentives
Financial incentives involve the creation of funds used for environmental improvement. Funds may come directly from government grants, from specific taxes or from external ‘deals’ such as a debt-for-nature swap, Global Environment Facility incremental cost financing etc. Financial incentives are especially important for the Economies in Transition.

Tradable quotas and offsets
Tradable quotas can relate to emissions (tradable emission permits) or resources (tradable resource quotas). Offsets relate to bargains between several parties such that an emission reduction obligation in one location is offset by reducing emissions in another location. The credits may not be traded (joint implementation) or they may be traded (tradable emission credits).

- joint implementation (mainly CO_2 but not exclusively);
- tradable emission permits (SO_x in the USA, and limited use in Europe): auctioned / grandfathered;
- tradable water rights;
- tradable fishing quotas: auctioned / grandfathered, and
- tradable development rights (land is zoned, some of it for development and rights to that development then become tradable).

Voluntary agreements
Voluntary agreements involve an understanding, sometimes backed by legal requirements, between government and industry such that industry ‘self regulates’. Self regulation involves setting agreed environmental targets, leaving industry to determine its own means of achieving those targets, such to some overall broad agreement on mechanisms.

Information
Two forms of information provision are considered:

- labelling (labelling of environmental performance, resource content etc)
- disclosure (publication of pollution profile of companies etc)

Matching policies to environmental problems
The following matrices ‘match’ environmental problems and economic instruments based on the five criteria indicated above. The allocation is necessarily judgmental but conforms with exercises elsewhere that have attempted to link instruments and problems in various different contexts. Environmental funds are excluded from the analysis because they can be created through the other instruments. However, externally financed funds are of importance to ‘economies in transition. Property rights are also excluded as they are of less concern in Europe. Nonetheless, each environmental problem is prefaced with a remark about property rights. These matrices are the foundations for the policy packages / assessments for each environmental issue presented in this Annex.

Stratospheric ozone depletion

Property rights in the ozone layer were established by the Montreal Protocol and the subsequent amendments and agreements. Rights are held by all signatory countries. Policy measures relate to controls on domestic production and controls on imports due to the fact that imported ODSs contribute to domestic consumption totals which, are the subject of 'caps'. Imports of recycled ODSs do not count against domestic consumption. *Financial incentives* (not shown here) relate to the Multilateral Fund which finances phase-out in the developing countries. Note also that the Montreal Protocol makes extensive use of restrictions on international trade in CFCs.

Stratospheric ozone depletion

Issue: reduce emissions of ODSs	
Initiatives	
Fiscal incentives	CFC taxes exist in USA Import duty reductions for ODS
Charges	-
Deposit refund schemes and performance bonds	DRS for recycled ODSs
Liability	-
Tradable permits	Reduction in trading in USA Permit trading for import allowances
Voluntary agreements	VAs to restrict imports
Information	Labelling products

Climate change

Property rights established by FCCC, 1992 and Kyoto Protocol 1997/8. Financing for LDC emissions reduction takes place via incremental cost financing from the Global Environment Facility, and through joint implementation schemes. The Clean Development Mechanism introduced in the Kyoto Protocol could evolve into a North-South JI scheme. JI East-West is enabled under the Kyoto Protocol.

Climate change

Issue reduce emissions of GHGs and sequestering carbon	
Initiatives	
Fiscal incentives	Carbon / energy taxes in place in several countries; Excise duties; Aviation tax; Methane tax;
Charges	-
Deposit refund schemes and performance bonds	-
Liability	-
Tradable permits	Joint implementation in place: over 200 deals Tradable efficiency permits for car manufacturers
Voluntary agreements	Carbon neutral pricing schemes Voluntary offset schemes
Information	Energy conservation campaign Emission disclosure

Major accidents

Major accidents occurring in a single Member State and not affecting other States need to be distinguished from accidents with potential transboundary effects. Nuclear, oil sill and chemical risks can easily be transboundary, suggesting that preventive and emergency response measures should be co-ordinated at EU level. To be realistic, such measures need to incorporate funds, akin to the Montreal Protocol Multilateral Fund, to finance risk reduction in the EITs. Such a fund exists for nuclear accidents (EU, Canada and US financed) and there are emergency response communications co-ordinated across Europe.

Major accidents

Issues reducing high risk nuclear reactors, reducing chance and impact of chemical disasters	
Initiatives	
Fiscal incentives	Tax on energy output Tax on port calls Output tax All to fund emergency responses
Charges	-
Deposit refund schemes and performance bonds	Could introduce performance bonds in EU
Liability	Negligence liability in EU
Tradable permits	-
Voluntary agreements	-
Information	-

Biodiversity loss

Unless privately owned or legally protected, most biodiversity is not the subject of property rights. Ownership of land by conservation groups or the state can contribute substantially to reducing biodiversity loss. Effectively, a market in biodiversity is created, although the medium is the land and property market itself. In other cases, market creation may be direct, e.g. by commercialising products from wild species, creating an incentive to conserve the species for profit. Since pollution is a cause of biodiversity loss it should be noted that all pollution reduction measures (see other environmental problems) will have an impact on biodiversity.

Biodiversity loss

Issue: reducing biodiversity loss through reduced habitat loss	
Initiatives	
Fiscal incentives	Agri-environmental schemes; environmentally sensitive areas, country side access schemes, country side stewardship schemes, Arable stewardship schemes, habitat schemes, moorland schemes, organic farming schemes, nitrate sensitive areas, Payments for set-aside; Outright land purchases; Tax allowances on money and land donations to conservation; Tax allowances on reforesting, soil and water conservation, and Easements and purchase of development rights;
Charges	Park entrance fees, user permits with earmaked revenues; Fines for damage to natural assets.
Deposit refund schemes and performance bonds	Land restoration with performance bonds
Liability	Liability for pollution damage
Tradable permits	Offset requirements, e.g. loss of wetland has to be offset by creation of new wetlands, i.e. mitigation banking; Tradable development rights; Tradable fishing quotas.
Voluntary agreements	Voluntary management agreements: Sweden, Austria, UK.
Information	Ecolabelling

Acidification and eutrophcation

Property rights to transboundary pollution reduction have been established by the Convention on Long Range Transport of Air Pollution in Europe and by various EU legislation.

Acidification and eutrophication

Issue: reducing emissions of SO_x, NO_x and NH_3	
Initiatives	
Fiscal incentives	S and N taxes NH_3 tax with mineral accounting, or livestock tax
Charges	-
Deposit refund schemes and performance bonds	-
Liability	-
Tradable permits	Possible tradable permits in SO_x and NO_x
Voluntary agreements	-
Information	-

Chemical risks

Chemical risks

Issue: reducing risks from heavy metals, pesticides and POPs	
Initiatives	
Fiscal incentives	Pesticide tax Battery charges Chemicals charges
Charges	-
Deposit refund schemes and performance bonds	Application to hazardous products, e.g. batteries
Liability	-
Tradable permits	Lead trading
Voluntary agreements	VAs possible
Information	Ecolabelling

Water management

Water management

Issue: improving water availability through management of supply and demand, and improving quality of ground water and surface water.	
Initiatives	
Fiscal incentives	Pesticide tax Fertiliser tax
Charges	Abstraction charges Effluent charges
Deposit refund schemes and performance bonds	-
Liability	-
Tradable permits	Tradable water rights Tradable effluent rights Tradable quotas for pesticides and fertilisers
Voluntary agreements	-
Information	-

Note: main requirement is to control for water demand through pricing of water at long run marginal cost.

Waste management

Waste management

Issue: reducing waste at source, increase recyling and re-use, minimise landfill	
Initiatives	
Fiscal incentives	Recycling credits; Virgin materials tax; Landfill tax; Incinertion tax.
Charges	Collection charges
Deposit refund schemes and performance bonds	DRSs for returnable containers; DRSs for batteries.
Liability	-
Tradable permits	Tradable recycling quotas
Voluntary agreements	Producer responsibility agreements
Information	-

Tropospheric ozone

Many of the policies suitable for acidification will also have a significant impact on the problem of tropospheric ozone.

Tropospheric ozone

Issue: reduce NO_x and VOC emissions (i.e. the precursors to low level ozone)	
Initiatives	
Fiscal incentives	N tax VOC tax
Charges	-
Deposit refund schemes and performance bonds	-
Liability	-
Tradable permits	Tradable quotas in VOCs and NO_x
Voluntary agreements	-
Information	Eco-labelling for solvents

Coastal zone management

See also climate change, biodiversity loss and chemical risks.

Coastal zone management

Issue: reduce coastal erosion (see climate change), reduce habitats damage (see biodiversity loss), improve bathing water quality.	
Initiatives	
Fiscal incentives for: Bathing water quality	 Tax non compliance with Bathing Water Directive
Charges	Possible beach charges
Deposit refund schemes and performance bonds	-
Liability	Owner liability for failure to meet Bathing Water Directive; Owner liability and performance bonds against oil spills.
Tradable permits	Transferable development rights; Tradable quotas for fishing
Voluntary agreements	-
Information	-

Human health, air quality and noise

Human health, air quality and noise

Issue: reduce exposure to noise, reduce urban pollutants especially PM10, PM2.5.	
Initiatives	
Fiscal incentives	Air pollution taxes: see acidification Noise taxes for vehicles Airport landing charges varied with noise levels;
Charges	Road user charges according to noise levels and congestion,
Deposit refund schemes and performance bonds	-
Liability	-
Tradable permits	Tradable efficiency permits for car manufacturers
Voluntary agreements	-
Information	-

Soil degradation

Soil degradation

Issue: reduce soil degradation from all causes but especially from water erosion	
Initiatives	
Fiscal incentives	Tax offsite damages Subsidies to good practice
Charges	-
Deposit refund schemes and performance bonds	-
Liability	-
Tradable permits	-
Voluntary agreements	Management agreements
Information	Extension services

Structure for policy packages

A policy package paper exists for each environmental problem. The structure of the policy packages is based on the following format:

1. *Key issues* associated with each environmental problem are described. This may include a comment about the expected benefits from environmental control as well as the most suited policies based on the five criteria;

2. *Available instruments*: based on the matrices that ‘match’ policies to environmental problems presented above and guided by the results of the ‘policy assessment’, the policy packages give more detail to the recommend policies. The section also provides *experience with policy instruments* in the EU15 (and elsewhere if relevant) and where possible an indication of the *effectiveness of the policy instruments* is given, i.e. a summary of what is known about the effectiveness of actual instruments, including simulations of hypothetical instruments and judgements.

Structure of policy assessments

This section assesses the suggested policies against the five criteria described above, i.e.

i) causal criterion,
ii) efficiency criterion,
iii) administrative complexity,
iv) equity criterion, and
v) jurisdictional criterion.

3. MONETARY VALUATION TECHNIQUES

Introduction

The economic approach to valuing environmental changes is based on people's preferences for changes in the state of their environment. Environmental resources typically provide goods and services for which there are either no apparent markets or very imperfect markets, but which nevertheless can be important influences on people's well-being. Examples include the quality of air, which affects people's health, crop yields, damage to buildings, and acidification of forests and fresh waters.

However, the lack of markets for these services means that unlike man-made products, they are not priced, therefore their monetary values to people cannot be readily observed. The underlying principle for economic valuation of environmental resources, just as for man-made products, is that people's *willingness to pay* (WTP) for an environmental benefit, or conversely, their *willingness to accept compensation* (WTA) for environmental degradation, is the appropriate basis for valuation.

If these quantities can be measured, then economic valuation allows environmental impacts to be compared on the same basis as financial costs and benefits of the different scenarios for environmental pollution control. This then permits an evaluation of the net social costs and benefits of each scenario for the different environmental issue.

The lack of markets and prices for many environmental goods and services means that the challenge for economists is twofold. The first task is to *identify* the ways in which an environmental change affects well-being; this is addressed in the next section, where the components of 'total economic value' of a resource are explained. The second task is to *estimate the value of these changes* through a variety of direct and indirect valuation techniques, exposition of which is given in the following sections.

Total Economic Value

The monetary measure of the change in society's well-being due to a change in environmental assets or quality is called the total economic value (TEV) of the change. To account for the fact that a given environmental resource provides a variety of services to society, TEV can be disaggregated to consider the effects of changes on all aspects of well-being influenced by the existence of the resource.

TEV can be divided into *use values* and *non-use values*, the latter also being called 'passive use values'. Use values include:

- direct use values, where individuals make actual use of a resource for either commercial purposes (e.g. - harvesting timber from a forest) or recreation (e.g. - swimming in a lake)
- indirect use values, where society benefits from ecosystem functions (for example, watershed protection or carbon sequestration by forests)
- option values, where individuals are willing to pay for the option of using a resource in the future (for example, future visits to a wilderness area)

Non-use values can take the form of:

- existence values, which reflect the fact that people value resources for 'moral' or 'altruistic' reasons, unrelated to current or future use
- bequest values, which measure people's willingness to pay to ensure their heirs will be able to use a resource in the future

Typically it is not possible to separate existence and bequest values.

To arrive at an estimate of the net change in societal well-being arising from an environmental change, we must consider each of these elements in turn. The total economic value (TEV) of a change is the sum of both use and non-use values:

TEV = use values + non-use values

= direct use + indirect use + option + existence + bequest values

Table 2.1 presents a taxonomy for environmental resource valuation, using the total economic value of a forest as an illustration.

Table 2.1 Economic taxonomy for environmental resource valuation

Total Economic Value				
Use Values			Non-use Values	
Direct Use	Indirect Use	Option Value	Bequest Value	Existence Value
Outputs directly consumable	Functional benefits	Future direct and indirect values	Use and non-use value of environmental legacy	value from knowledge of continued existence
• food • biomass • recreation • health	• flood control • storm protection • nutrient cycles	• biodiversity • conserved habitats	• habitats • prevention of irreversible change	• habitats • species • genetic • ecosystem

The first step in estimating any of these values is the definition and measurement of the environmental problem. This often includes an element of scientific uncertainty that can, at times, be quite significant. The accuracy of economic valuation is therefore dependent on accurate scientific identification and quantification of the environmental change in order to estimate people's preferences for or against it.

Valuation Techniques

The practical problem with economic valuation is one of deriving credible estimates of people's values in contexts where there are either no apparent markets, or very imperfect markets. In the case of marketed goods, price is the measure of willingness to pay and can be readily observed. However, in the case of non-marketed goods and services we need to elicit this value in different ways. There are two broad approaches to valuation, each comprising several different techniques, as illustrated in Figure 2.1.

- **Revealed preference techniques,** which infer preferences from actual, observed, market-based information. Preferences for environmental goods are revealed indirectly when individuals purchase marketed goods which are related to the environmental good in some way.

- **Stated preference techniques,** which attempt to elicit preferences directly by use of questionnaire, such as contingent valuation. All valuation of non-use values depends on these techniques.

We consider each of these approaches in turn, highlighting when each could be used, their advantages and drawbacks and their applicability to waste management problems.

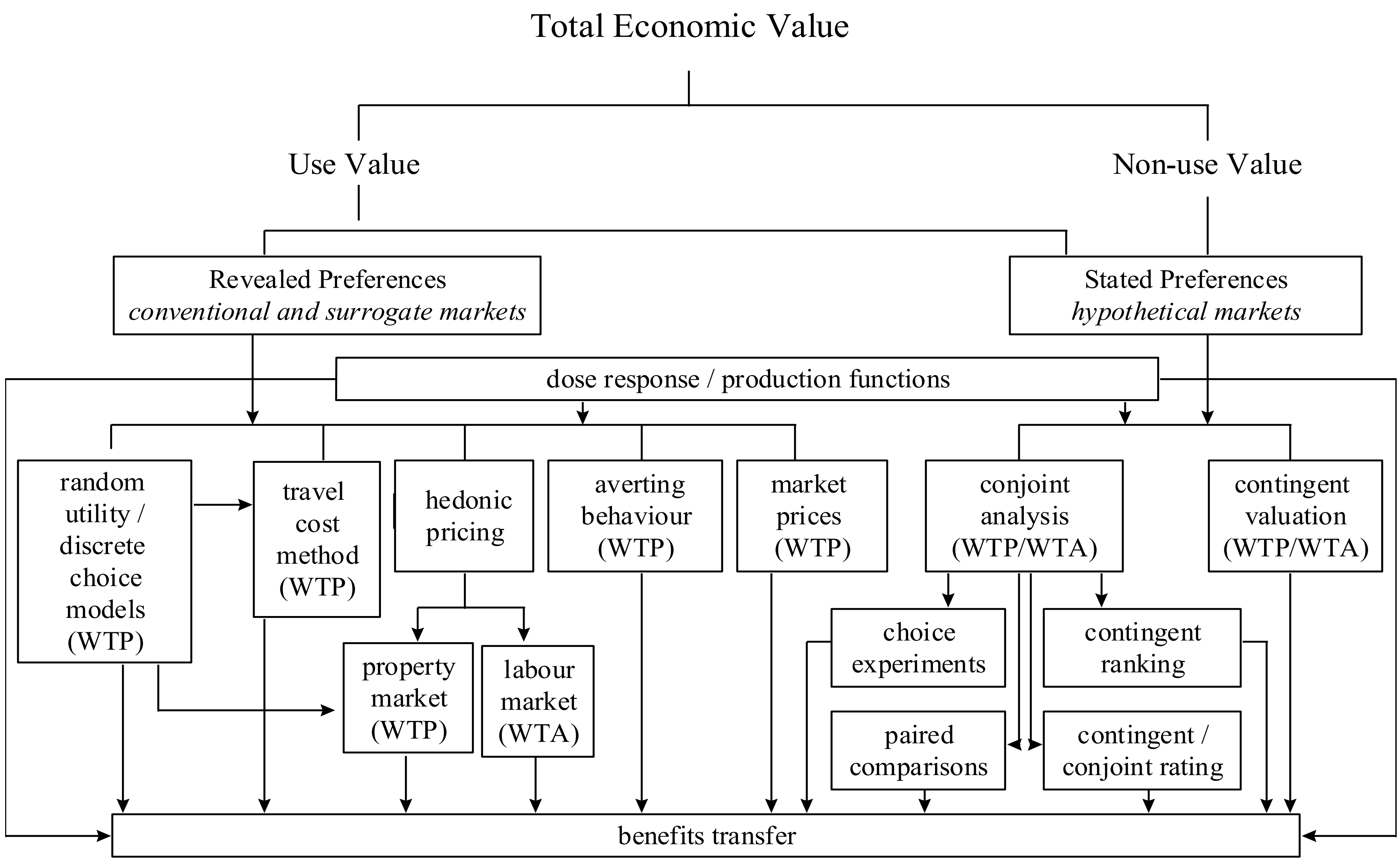

Figure 2.1: A typology of monetary valuation methods

Revealed Preference Techniques

The essence of revealed preference techniques is that they infer environmental values from markets in which environmental factors have an influence. For example, there are markets for certain goods to which environmental commodities are related, as either substitutes or complements to the goods in question. In this way people's actions in actual markets reflect, to a certain extent, their preferences for environmental assets.

There are four main revealed preference techniques that are considered in the sections that follow.

1. Averting behaviour
2. Hedonic pricing (of property and labour)
3. Travel cost method
4. Random utility and discrete choice modelling

Averting Behaviour

The basis for the averting behaviour technique is the observation that marketed goods can act as substitutes for environmental goods in certain circumstances. When a decline in environmental quality occurs, expenditures can be made to mitigate the effects and protect the household from welfare reductions. For instance, expenditure on sound insulation can indicate households' valuation of noise reduction; expenditure on household water filters can be used to estimate economic values of clean water.

The method is applicable in situations where households spend money to offset environmental impacts. It requires data on the environmental change and its associated substitution effects. Fairly crude approximations can be found by simply looking directly at changes in expenditures on the substitute good resulting from some environmental change.

Advantages of these models are that they have relatively modest data requirements and can provide theoretically sound estimates based on actual expenditures. However, they can give incorrect estimates if other important aspects of individuals' behavioural responses are ignored. For example, individuals may engage in more than one form of averting behaviour in response to any one environmental change. Additionally, the averting behaviour may have other beneficial effects that are not considered explicitly, for example sound insulation may also reduce heat loss from a home. Furthermore, averting behaviour is often not a continuous decision but a discrete one: for example, a smoke alarm is either purchased or not. In this case the technique will tend to underestimate the value of the environmental good.

Hedonic Pricing

This technique depends on analysis of existing markets where environmental factors have an influence on price. The example most frequently used is that of the housing market, as the environmental attributes of a property will vary according to its location. For example, noise levels will be higher close to an airport and, other characteristics being equal, this can be expected to lower the price of a property in the area. Similarly, two identical properties which differ only in, say, the local air quality, will differ in value to the extent that people find one air quality preferable to the other. The difference can be viewed as the value attached to the difference in air quality as measured by willingness to pay (WTP).

The hedonic property price (HPP) method can be used even when properties differ in many factors other than environmental quality provided that data are detailed enough. With the use of appropriate statistical techniques, the hedonic approach attempts to (i) identify how much of a price differential is due to a particular environmental difference between properties, and (ii) infer how much people are willing to pay for an improvement in environmental quality that they face and what the social value of the improvement is. The same technique has also been applied to labour in the valuation of work-related risk in hedonic wage (HW) studies. Identification of wage differentials due to differences in safety risks, for example, will give an indication of willingness to accept compensation (WTA) for incurring these risks, which can be used as a measure of the benefits of improving safety.

This technique can in theory be used to estimate factors such as the disamenity costs of location near to landfill sites, or air quality near to incinerators.

Travel Cost Method

Many natural resources are used extensively for the purpose of recreation. It is often difficult, however, to value these resources because no prices generally exist for them. The travel cost approach is based on the fact that, in many cases, a trip to a recreational site requires an individual to incur costs in terms of travel, entry fees, on-site expenditures and time. These costs of consuming the services of the environmental asset are used as a proxy for the value of the recreation site and changes in its quality.

Clearly, because travel cost models are concerned with active participation they measure only the use value associated with any recreation site. The method is now well -established as a technique for valuing the non-market benefits of outdoor recreation resources. It is useful because it is based on actual observed behaviour. However, the technical and data requirements are such that it is not readily applicable.

Random Utility or Discrete Choice Models

While the travel cost method is useful for measuring total demand or WTP for a recreational site, this technique is less useful for estimating the value of particular features or assets of the site which may be of interest. Random utility models have been developed for this purpose.

The emphasis of random utility or ‘discrete choice’ models is on explaining the choice between two or more goods with varying environmental attributes as a function of their characteristics. This can be useful where, for example, polluting activity causes damage to some features of a recreational site but leaves others relatively unharmed.

This can be illustrated using a simple example from a choice of transport mode. Supposing that, when undertaking a given journey, an individual faces the choice of travelling by taxi or by public transport. A taxi will take 20 minutes and cost €5, whereas public transport will take an hour but cost € 2. If the individual chooses to travel by taxi, it can be inferred that s/he judges the difference of 40 minutes in time to be worth at least the € 3 difference in fare. In other words, the value of the individual’s time is at least €4.50 per hour.

Another example is the choice between bottled water and tap water for drinking. The former is more expensive but associated with better quality. Therefore, the price difference between bottled and tap water is an indication of the value of risk in this context.

Replacement Cost

The replacement cost technique uses the cost of replacing or restoring a damaged asset to its original state as the measure of the benefit of restoration. The approach is widely used, largely because it is comparatively easy to apply.

This approach is valid where it is possible to argue that the remedial work must take place because of some other constraint such as an environmental standard. Replacement will only be efficient, however, if the environmental standard itself is economically efficient. Otherwise the approach estimates only the costs of replacement: it is not a technique for *benefit* estimation. Indeed, if costs of replacement are used to estimate the benefits of replacement, then a benefit-cost ratio of one will always result, and a replacement project will always appear justified.

Information on replacement costs can be obtained from direct observation of actual expenditure on restoring damaged assets or from engineering estimates of restoration costs. The technique implies various assumptions, for instance, that complete replacement is, in fact, feasible. In general, however, the potential for confusion between costs and benefits means that the replacement cost technique needs to be applied with some care.

Stated Preference Techniques

Stated preference techniques enable economic values to be estimated for a wide range of commodities which are not traded in markets. In addition, these techniques are the only way to estimate non-use value of environmental resources. Here, we consider two approaches:

1. Contingent valuation
2. Conjoint analysis

Contingent Valuation

In contingent valuation (CV) studies, people are asked directly to state what they are willing to pay for a benefit or to avoid a cost, or, conversely, what they are willing to accept to forego a benefit or tolerate a cost. A contingent market defines the good itself, the institutional context in which it would be provided, and the way it would be financed. The situation the respondent is asked to value is hypothetical (hence, 'contingent') although respondents are assumed to behave as though they were in a real market. Structured questions and various forms of 'bidding game' can be devised to assess the maximum willingness to pay. Econometric techniques are then applied to the survey results to derive the average bid value, i.e. the average WTP.

There are three basic parts to most CV surveys. First, a hypothetical description of the terms under which the good or service is to be offered is presented to the respondent. Information is provided on the quality and reliability of provision, timing and logistics, and the method of payment. Second, the respondent is asked questions to determine how much s/he would value a good or service if confronted with the opportunity to obtain it under the specified terms and conditions. These questions take the form of asking how much an individual is willing to pay for some change in provision. Respondents are reminded of the need to make compensating adjustments in other types of expenditure to accommodate this additional financial transaction. Econometric models are then used to infer WTP for or WTA the change. Finally, questions about the socio-economic and demographic characteristics of the respondent are asked in order to relate the answers respondents give to the valuation question to other characteristics of the respondent, and to those of the policy-relevant population.

CV is likely to be most reliable for valuing environmental gains, particularly when familiar goods are considered, such as local recreational amenities. While the accuracy of results also depends on careful construction of the survey, a set of guidelines for applying CV to derive reliable estimates of non-use values is developed by the US National Oceanic and Atmospheric Administration (NOAA) panel (Arrow et al.,, 1993). This is now being extended to cover all CV studies.

Conjoint Analysis

Conjoint analysis (CA) is a broad term used to cover several different techniques, all of which are survey methods, but they involve asking individuals to rank alternatives rather than explicitly express a WTP or WTA. For contingent ranking, the inclusion of prices in some of the alternatives enables rankings to be converted to monetary values. Other aspects are similar to contingent valuation.

Again, the main application of relevance to the current study has been in the context of human health and landscape effects, as well as disamenity.

Dose- and Exposure-Response Functions

Dose-response functions (DRFs) measure the relationship between a unit concentration of a pollutant and its impact on the relevant receptor. Exposure-response functions (ERFs) are based on the same principle but measure the response with respect to the exposure. Exposure is a measure of the levels of a pollutant in the environment surrounding the receptor in question. For example, a person may be exposed to a certain concentration of an atmospheric pollutant, but the dose received will depend on the amount inhaled, which is higher during exercise and lower during rest. In general, effects will be more closely related to dose, but it is much easier to measure exposure. Hence it is important to recognise that any dose-response function is often represented by the approximation of an exposure-response function (ApSimon et. Al., 1997).

Dose-response techniques are used extensively where a physical relationship between some cause of damage, such as pollution, and an environmental impact or ‘response’ is known and can be measured. Once the relationship has been estimated, then WTP measures derived from either conventional market prices (which are adjusted if markets are not efficient) or revealed/inferred prices (where no markets exist) using one of the techniques described in the previous section. The physical damage is multiplied by this shadow price, or value per unit of physical damage, to give a ‘monetary damage function’.

The approach is theoretically sound, and can be used wherever the physical and ecological relationships between a pollutant and its output or impact are known. The specification of the D/ERF is crucial to the accuracy of this technique, and is the main source of uncertainty. Difficulties and uncertainties may arise in: identifying the pollutant responsible for the damage and all possible variables affected; isolating the effects of different causes to determine the impact on a receptor, e.g. synergistic effects where several pollutants or sources exist; identification of damage threshold levels and the long term effects of low to medium levels of pollution. All these problems make it difficult to determine the appropriate empirical specification of the functional form. Additionally, there is the further complication that evidence of a physical response may not be economically relevant if individuals are not concerned about it and, therefore, do not attach a value to avoiding it. For these reasons, large quantities of data may be required and the approach may be costly to undertake.

If, however, the D/ERFs already exist and the impacts are marginal, the method can be very inexpensive and provide reasonable first approximations to the true economic value measures.

4. BENEFITS TRANSFER

The benefit assessment procedure conducted in this study makes extensive use of 'benefits transfer', i.e. taking existing monetary valuation (willingness to pay) studies and applying them outside of the site contexts where the study was originally carried out. There is in fact no alternative to this procedure if any use at all is to be made of benefit valuation techniques. The approach implicitly underlies the procedures used, for example, by ExternE, although, as it happens, their use of transferred functions may be more basic than ours in at least one respect. They choose specific functions and apply these across Europe. In our case we make some attempt to engage in 'meta studies' where that is possible. Technically, the alternative is to carry out willingness to pay studies across all EU15 countries for all environmental problems. Clearly, this is not possible. Nonetheless, we should be aware that the procedure involves risks and errors. This note serves to set out the nature and problems involved in benefits transfer. It should be noted that the literature analysing the validity of transfer techniques is very small.

(a) *Transferring average WTP from a single study to another site which has no study*

The basic idea is to 'borrow' an estimate of WTP in context i and apply it to context j, but making adjustments for the different features of the two contexts. For example, if incomes vary we might have

$$WTP_j = WTP_i(Y_j/Y_i)^e$$

where Y is income per capita, WTP is willingness to pay, and 'e' is the income elasticity of demand, i is usually called the *study* site and j the *policy* site.

A typical example of such an approach is given by Krupnick et al., (1996) who transfer US WTP for various health states to Eastern Europe using the ratio of wages in the two areas and an income elasticity of demand of 0.035. The significance of the procedure can be realised since the wage ratio raised to e=0.035 produces a WTP in Eastern Europe equal to only 8% of that in the USA.

A second, common adjustment is for population size and, less frequently, for the distribution of population characteristics, e.g. age.

Note that the transfer is 'assumed' to be correct: no separate validation is carried out. This is similar to most of the transfer of values used in the EU Priorities study.

(b) *Testing the equality of means at two sites where studies exist*

Where there are two sites both with actual WTP estimates we can obtain some idea of the validity of benefits transfers by comparing the two mean WTPs. We wish to know if they are statistically the same. If they are, then there is some reason to feel confident that the results from a given site can be transferred to another site, as in (a) above.

Where the underlying distribution of WTP is thought to be normal, parametric tests can be used (eg t-tests) to determine if the mean WTP results at the two (or more) sites are statistically the same. Where this restriction is thought to be unreasonable, then non-parametric tests are required. More sophisticated testing can be done, e.g. to find out if the two underlying WTP distribution (not just the means) are statistically the same.

(c) *Transferring benefit functions*

A more sophisticated approach is to transfer the *benefit function* from i and apply it to j. Thus if we know that WTPi = F(A, B, C, Y) where A,B,C are factors affecting WTP at site i, then we can estimate WTPj using the coefficients from this equation but using the values of A, B, C, Y at site j.

Alternatively, we can use *meta analysis* to take the results from a number of studies and analyse them in such a way that the variations in WTP found in those studies can be explained. This should enable better transfer of values since we can find out what WTP depends on. Whole functions are transferred rather than average values, but the functions do not come from the single site i, but from a collections of studies.

(d) *Is transferring functions valid?*

How do we know if transferring functions is a valid procedure? As with the procedure under (a), we have no direct test that the result is ‘correct’. The literature has proceeded by taking estimated demand functions at site i and site j and then comparing them to see if, statistically, they are the same. This involves at least testing for the equivalence of the coefficients in the two functions, e.g.

$$WTP_i = x + a_1 A + b_1 B + c_1 C$$

and $$WTP_j = x + a_2 A + b_2 B + c_2 C$$

so that we require $a_1 = a_2$ etc, where equality here is statistical equality (Loomis, 1992).

Recent literature has suggested that even if it is valid to transfer *benefit functions*, based on statistical equality of coefficients, the resulting estimates of *benefits* may be in error. This is because benefits may not be a linear function of the coefficients. Downing and Ozuna (1996) take demand functions for 8 sites in Texas and conclude that around 50% of functions are transferable (have the same coefficients) but that only a small minority would yield reliable benefit estimates. This has led Bergland et al., (1995) to suggest that both valuation functions and benefits estimates must be transferable (see the ‘protocol’ below..)

Generally, the literature testifies to the unreliability of transferring benefit functions (Loomis, 1992; Downing and Ozuna, 1996; Bergland et al.„ 1995; Parsons and Kealy, 1994). Most studies seem to suggest that transferring functions is better than transferring average values, but that both are subject to significant margins of error (Kirchhoff et al.„ 1997).

(e) *Validating benefits transfer*

The test in (d) above involves taking actual demand functions and seeing whether they are statistically the same and will produce similar benefit estimates. Another test would be to take a WTP estimate from i and apply it to j using a simple procedure such as the one set out in (a) above. Then, a full WTP study would be carried out in j and the mean WTP result would be compared with the ‘transferred’ WTP.

Navrud (1997) has done this for minor impaired health states to see if WTP estimates from the USA can be transferred to Europe (in fact, to Norway). He concludes that the transferred estimates significantly overstate the ‘actual’ WTP as derived from a contingent valuation study in Norway.

Alberini et al., (1995) make this test of benefits transfer using two US contingent valuation studies of a 'restricted activity day' due to a head cold and transferring the results to Taiwan. In this case the transfer multiplier was (Yj/Yi) which implies e=1.

They then carried out a contingent valuation of the morbidity effect in Taiwan. The results were statistically the same, ie the simple benefits transfer approach accurately predicts the policy site study results.

(f) *The Bergland-Magnussen-Navrud Protocol*

Bergland et al., (1995) (BMN) recommend testing for benefits transfer in four stages:

1 test that mean $WTP_i = WTP_j$, using parametric and non-parametric tests depending on the assumed underlying distribution of WTP

2 estimate WTP'_j where WTP'_j uses estimated parameters from i and the actual values of explanatory variables at j. Test for the equivalence of $WTP'_j = WTP_j$, ie we require

$$WTP'_j = f(b_i, X_j) = WTP_j$$

and correspondingly for WTP'_i.

3 compare parameters b in each study, with the requirement that

$$b'_i = b_j \text{ and}$$

$$b'_j = b_i$$

where b'_j comes from estimating the function $WTP'_j = f(b_i, X_j)$ above, and correspondingly for b'_i.

4 test for the proposition that the two benefit functions come from one underlying function with parameters b such that

$$b = b_j = b_i.$$

Criteria for successful Benefits Transfer

It appears generally agreed that successful benefits transfer requires:

1 adequate data for those studies included in the analysis
2 sound economic and statistical technique
3 studies with regressions of WTP on determining variables
4 similar populations in the compared sites
5 similarity of the environmental good to be valued
6 similar sites
7 similar distributions of property rights.

See, for example, Brouwer and Spaninks (1997).

Conclusions

The literature on benefits transfer is very small. The attractions of benefits transfer are very clear: without it, one has to resort to primary valuation studies. This is both expensive and time consuming. It would not matter for 'micro' problems where it is often possible to carry out such studies, but it is a problem for wide-ranging studies such as the European Environmental Priorities study where we require valuations across many Environmental problems, across the EU 15 countries and, where possible, across Accession countries.

At the moment, the literature reports mixed results with the balance of opinion expressing considerable caution about benefits transfer. It seems clear that the conditions required for 'good' transfer are not met in the kind of the analysis where single estimates are applied across man countries. The error is likely to be reduced substantially wherever meta analysis can be done and meta-functions can be applied. Even here, there are some doubts about the validity of transfer.

5. INCOME ELASTICITY OF DEMAND FOR THE ENVIRONMENT

Benefits Transfer and the Income Elasticity of Demand

Finding 'unit values' for changes in water quality or water availability rests heavily on benefits transfer, i.e. the process of taking values from one context and applying them to another. Benefits transfer usually involves adjustments to the original estimates, adjustments that can be quite complex but which are often very simple. A formula that is quite widely used but which adjusts only for differences in income between the original site (i) and the site to which the estimate is to be transferred (j) is:

$$B_j = B_i.(Y_j/Y_i)^e \qquad ...[1]$$

where 'B' is benefit (measured by willingness to pay), 'Y' is income, and 'e' is the 'income elasticity of environmental value'.

Here we focus on how to find values of 'e'.

The Analytics

There are two parameters that are relevant in estimating the income elasticity of demand for the environment.

The first is the conventional measure of *income elasticity of demand*

$$\eta = \Delta X.Y/\Delta Y.X \qquad ...[2]$$

where, Δ is change in, X is the quantity of the environmental good in question and Y is income.

Traditionally, goods have been classified in terms of the value of η:

Value of η	Share of expenditure on good X as Y rises	Name of good
$\eta <0$	Falls	inferior
$0 < \eta > 1$	Falls	normal, necessity
$\eta = 1$	Constant	normal
$\eta > 1$	Rises	normal, luxury

Casual commentators have often argued that the environment is a *luxury good*, i.e. that it is something that societies worry about only when incomes rise.

The second indicator is the *income elasticity of environmental valuation*, or willingness to pay (WTP):

$$e = \Delta WTP.Y/\Delta Y.WTP \qquad ...[3]$$

The relationship between [2] and [3] is easily derived. From [3] we have

$$Y/\Delta Y = e.X/\Delta X$$

and substituting in [3] gives:

$$e = \eta.\Delta WTP.X/WTP.\Delta X \qquad [4]$$

Note that $e < 1$ is quite compatible with $\eta > 1$, so that a good that is a 'luxury good' can have an income elasticity of WTP < 1 (Flores and Carson, 1997).

Which is the relevant concept ? Since the focus of most environmental policy is on public goods that have some quantity constraint, it turns out that it is the second concept - the income elasticity of environmental values - that is more relevant.

Empirical evidence (a) environmental goods
In an early survey, Pearce (1980) assembled what evidence there was on income elasticities of WTP for the environment. He found (a) that hedonic property price models could not be used to *infer* income elasticities because the models themselves tended to constrain the values to be unity anyway, and (b) what evidence there was suggested that income elasticities were *less than unity.* Kristrom and Riera (1996) review more recent evidence and reach a similar conclusion. Analysing six European contingent valuation studies, Kristrom and Riera find that the share of expenditure on environment falls as income rises, i.e. $e < 1$. This result is supported with evidence from Australian and US CVM studies, with other work from Africa and the USA, and with analysis of corporate donations to environmental causes in the USA.

Empirical evidence (b) risks
Values of $e < 1$ have also been obtained from studies of valuations of statistical life. These produce a range of values of 0.3 to 1.1 with the majority of estimates being at the lower end of the range, ie well below unity:

Blomquist, 1979	e = 0.3
Jones-Lee et al., 1985	e = 0.3
Persson and Cedervall, 1991	e = 0.3
Jones-Lee et al., 1993	e = 0.3
Miller and Guria, 1991	e = 0.3 to 0.6
Persson et al., 1995	e = 0.46
Viscusi and Evans, 1990	e = 1.0 (non fatal injuries)
Kidholm, 1994	e = 1.1

(taken from a survey by NERA, 1997).

While the number of studies remains limited, it is difficult to avoid the conclusion that the environment is *not* a luxury good on the relevant definition of income elasticity of WTP.

6. VALUING STATISTICAL LIVES

Introduction: the Importance of Risk Valuation in Environmental Cost-Benefit Studies

Environmental cost-benefit studies include as benefits any reductions in the risks of premature mortality and morbidity. In turn, changes in the risks of health 'end points' are given economic valuations based on the willingness to pay (WTP) of those at risk to reduce the risks. Valuations may vary with the level of risk and certainly vary with the health state that is avoided, e.g. people are more averse to cancer risks than risks of accident. One feature of these cost benefit studies is that health benefits tend to dominate overall benefit estimates. Accordingly, if the basis on which the health benefits are estimated is incorrect, then the overall cost-benefit result is very likely to be incorrect. It matters a great deal, therefore, if the underlying epidemiology is correct and if the economic valuation applied to the health effects is correct.

This paper provides an overview of the issues as they relate to premature mortality only. It is designed as a background paper on the debate about the appropriate way to treat life risks in the context of environmental change. It does not seek to produce any new results, being designed mainly for reference and as a guide to the issues.

Table 5.1 shows the role that health benefit valuation has played in some recent European cost-benefit studies. It can be seen that the overall benefits figures are dominated by health impacts. Other studies report cost-benefit results for policies aimed directly at health effects. Here the issue is whether benefits exceed costs, an issue that is also very much affected by the approach taken to health impacts.

Table 5. Health benefits as a percentage of overall benefits in recent cost-benefit studies

Study	Title and subject area	Health benefits as % of total benefits
Holland and Krewitt, 1996	*Benefits of an Acidification Strategy for the European Union*: reductions of SO_x, NO_x, NH_3 in the European Union	86-94%. Total benefits cover health, crops and materials.
AEA Technology, 1998a	*Cost Benefit Analysis of Proposals Under the UNECE Multi-Effect Protocol*: reductions of SO_x, NO_x, NH_3, VOCs	80-93%. Total benefits cover health, crops, buildings, forests, ecosystems, visibility
IVM, NILU and IIASA, 1997	*Economic Evaluation of Air Quality for Sulphur Dioxide, Nitrogen Dioxide, Fine and Suspended Particulate Matter and Lead:* reductions of these pollutants	32-98%. Total benefits include health and materials damage
AEA Technology, 1998b	*Economic Evaluation of the Control of Acidification and Ground Level Ozone*: reductions of NO_x and VOCs. SO_2 and NH_4 held constant.	52-85% depending on inclusion or not of chronic health benefits. Total benefits include health, crops, materials and visibility

Economic Valuation and Resource Allocation

Economic valuation is intricately inseparable from the issue of how to allocate scarce resources. Risk reduction is not a costless activity and hence any resources used up in the reduction of one set of risks could have been used to reduce another set of risks. Taking a wider view, resources allocated to risk reduction might equally be allocated to some entirely different purpose: education, restoring national heritage, improving landscapes, and so on.

Valuation attempts to provide the answer to the problem of choosing between alternative uses of resources. If risk reduction has a high value relative to other uses of resources, then risk reduction should have priority.

There have been philosophical objections to the use of economic valuation. On what might be called the 'rights approach' individuals have rights to human health and a clean environment, and such rights would have similar status to rights against discrimination (Bullard, 1994). One possible implication of the rights approach is that all environmental risks should be reduced to zero, since any positive level of risk infringes individuals' rights. Alternatively, if rights conflict and are not absolute, then some trade-off between rights has to occur. A variation of the rights-based approach extends rights to non-human species, i.e. it confers 'value' on living things, and sometimes non-living things independently of human values. This is sometimes articulated in terms of 'intrinsic' rights of species to exist.

The rights-based approach contrasts with the view based on 'trade offs' between cost and risk reduction. Risk reduction is pursued up to some point where the costs of such action are thought to be 'too high'. There are divergent views as to how this trade-off is to be made. In particular, there are those who favour a balancing of economically valued costs and benefits, and there are those who favour leaving the trade-off to the political system. This categorisation is not meant to be all -encompassing. More detail of the considerable variation of views within these categories can be found in Turner (1993).

One of the problems with the debate about these alternative views is that much of the discussion takes place quite independently of the real world context of environmental change. If resources were infinite there would be no problem of trade-off, and hence no problem of determining priorities. Everything deemed to be 'good' or 'right' could be done. But the real world is not like this and it is necessary to choose. The fundamental feature of choice making is cost, which is another way of saying that resources are finite. Adopting a rights-based approach implies that the choices surrendered by pursuing risk reduction as a matter of right (i.e. the cost) are of a lower 'moral order' than risks to human health or risks to other species. The problem then is that risk reduction has to be pursued regardless of the forgone values sacrificed. Moreover, all risk reduction has to be pursued: it cannot be correct to reduce some risks but not others unless the rights are attenuated in some way. Risk reduction may therefore conflict with other rights, e.g. rights to a decent livelihood, rights to education, and, especially, rights to freedom of choice.

Much of the motive for the rights-based approach arises from an understandable sense of frustration with the fact that trade-off approaches do involve 'acceptance' of some positive levels of risk. But it also has its foundations in a lack of appreciation of what 'cost' actually means, a perception fostered by the view that cost is 'just money', as if money is unrepresentative of human wellbeing.

Developments in risk analysis sharply underscore the unavoidability of trade-offs and the unreality of the rights-based approach. Risk-risk analysis and health-health analysis draw attention to the fact that the costs of risk reduction policies are met from reductions in household incomes - see Keeney (1990, 1994, 1997), Graham et al., (1992), Lutter and Morrall (1994), Portney and Stavins (1994), Viscusi (1994) and Viscusi and Zeckhauser (1994). It is known that households with low incomes tend to have higher exposure generally to life and health risks, so that reductions in expenditure increase risk exposure. For example, Keeney (1997) estimates that in the USA there is one fatality for each $5-11 million of public expenditure on risk reduction.

Finally, rights-based approaches tend to be discussed as if whatever is deemed to be 'right' by one or more persons constrains others who may not share the moral view. Put another way, what is right has an absolutist flavour. If there were no dispute about the moral standpoint, then, clearly, there would be a moral consensus. But in so far as hypothetical market studies have shown the existence of 'lexical' preferences (implying no trade off) - and this is disputed - they have not characterised the whole sample. It is unclear therefore what role a minority believing in ‘rights’ should play in determining the outcome of a policy or project choice.

Overall, then, rights-based approaches fail because of their neglect of the most basic of all economic principles - opportunity cost, and because they have little to say about consensus.

The Trade -Off View: Economics

The economic approach to the trade-off issue operates through the aggregation of human preferences. The set of persons affected by a decision defines the set of people whose preferences count, where ‘affected by’ means that their wellbeing is, in one way or the other, partly dependent on the environment in question. This preference-base is inherently ‘democratic’ - it requires that policies be responsive to preferences however they are formed. Preferences are revealed in the market -place through demand behaviour -i.e. as ‘willingness to pay’ (WTP). Indeed, the demand curve in textbook economics is a (marginal) willingness to pay curve. If WTP is rejected as a criterion for allocating resources to risk reduction, then some explanation has to be provided as to why environmental goods and services are different to other goods and services which are allocated on a WTP basis.

But risk reduction often has no market, i.e. the issue giving rise to risk is not bought and sold on the open market. Clean air would be an example. Thus the economic approach requires that preferences for risk reduction be inferred from human behaviour in other contexts.

The theory of economic valuation has developed substantially in the last two decades. This section reviews, briefly, those techniques that relate to human health risks only. Other techniques are relevant to the valuation of other environmental changes. For a detailed review see Freeman (1993).

For a change in risk that threatens life and health generally, we can say that the relevant valuation is the value that the individual at risk attached to their own health and life chances, plus what others would be willing to pay to avoid the risk to that individual, plus any costs that society at large bears and which would not otherwise occur if the individual did not suffer the effects of the risk in question. These components of this value of risk (VOR) are:

(a) $VOR_{i,i}$ where i,i refers to the individual i’s valuation of risk to themselves, i.e. ‘own risk’. The way in which these individual VORs are aggregated is dealt with shortly. Essentially, we will require the summation of such own valuations for all individuals at risk to give $\Sigma_i VOR_{i,i}$, more commonly known as the ‘value of a statistical life’ - see below.

(b) $VOR_{i,j}$ where the i,j notation now refers to j’s valuation of risks to i. Again, this will need to be summed for all j, i.e. for all people expressing some concern about risks to i, to give $\Sigma_j VOR_{i,j}$.

(c) COI_i where COI refers to the ‘cost of illness’ suffered by i but which costs are borne by the rest of society. An example would be hospital costs. COI could be regarded as part of VORi,j

The extent to which these three components of the value of life risks can in fact be *aggregated* is discussed later.

Valuing Statistical Lives

One form of health risk is the risk of premature mortality arising from some risk context, say increased air pollution. What value should be attached to such risks of mortality? The sum of individuals' own valuations of risks to their own lives is known as the value of a statistical life, VOSL. The shorthand often used for the VOSL is 'value of life', which is unfortunate. Since the idea of 'valuing life' appears odd to some and morally offensive to others, it is important to understand what a value of a statistical life (VOSL) actually is.

The way a VOSL is obtained is by aggregating up from a value (willingness to pay, WTP) of risk reduction. Imagine the probability of dying next year is 0.004 for each person and suppose we have 1000 persons in the population. Assume there is some risk reduction policy that reduces the risk to 0.003, a change of 0.001. Each person is asked to express their WTP for this change in risk and suppose the answer is £1000. The risk reduction policy is a public good: it affects everyone equally. Thus 1000 people say they are each willing to pay £1000 for the policy, i.e. their aggregate willingness to pay is £1 million. The change in risk will result in one statistical person being saved each year (1000 x 0.001). Thus the value of a statistical life is £1 million in this example. It is important to understand that no one is being asked their WTP to avoid themselves dying at a specified time: they are being asked to express a WTP for a change in risk. As Freeman (1993) notes:

> '..the economic question being dealt with here is not about how much an individual would be willing to pay to avoid his or her certain death or how much compensation that individual would require to accept that death. In this respect, the term "value of life" is an unfortunate phrase that does not reflect the true nature of the question at hand. Most people would be willing to pay their total wealth to avoid certain death; and there is probably no finite sum of money that could compensate an individual for the sure loss of life. Rather, the economic question is about how much the individual would be willing to pay to achieve a small reduction in the probability of death during a given period or how much compensation that individual would require to accept a small increase in that probability.' (p320).

It is worth emphasising Freeman's point: the VOSL is *not* what someone is willing to pay to avoid losing their life, a confusion that is pervasive in the popular literature commenting on valuations of life risks. It is the valuation of small changes in risk. VOSL is essentially a convenient rule for aggregation.

Individuals' WTP to reduce risks can be expected to vary across different individuals. The two main reasons for this will be that:

> (a) people have differing attitudes to risk: some may even be 'risk lovers', i.e. positively enjoying risky contexts. Most people are risk avoiders, i.e. they will tend to reveal a positive willingness to pay for risk reduction. But there is no particular reason why their valuations of risk should be the same;
>
> (b) incomes vary and hence willingness to pay is likely to vary in such a way that those with higher incomes have higher WTPs. This is not a necessary result since attitudes to risk may vary in such a way as to offset an income effect. Nonetheless, it raises an important equity issue about fairness between people, an issue that is not in fact confined to risk valuations but to the use of WTP measures in general.

A VOSL can also be measured by a 'willingness to accept' compensation for increased risk. It is well known that many people do make this trade-off between risk and money, for example by accepting premia on wages to tolerate risk. It is tempting to think that the WTA approach will produce very much higher values for a VOSL than the WTP approach, simply because WTA is not constrained by income. WTP and WTA can, indeed, be different and WTA for environmental losses may exceed WTP for environmental gains by factors of 2-5 (Gregory, 1986). Various explanations exist for this disparity, including the fact that individuals may feel they are losing an 'entitlement' if the issue is one of loss of an entitlement (WTA) rather than an increment to an existing entitlement (WTP). Another explanation, which is wholly consistent with economic theory, suggests that WTA > WTP arises mainly in contexts where there is no ready substitute for the environmental good in question (Hanemann, 1991). These issues are discussed further later on.

Techniques for Estimating VOSL

A number of techniques have been developed to estimate VOSLs. The main ones are rooted in the general economic theory of valuation, i.e. they have a theoretical basis on the measurement of human wellbeing based on individuals' preferences. One widely used technique, however, has only a tenuous link to the theory.

Valuing Mortality Risks: Wage Risk Models

The wage risk, or 'hedonic wage' model estimates a willingness to accept measure of risk. Essentially, it looks at wages in risky occupations and seeks to determine the factors that determine wages. One of these factors is hypothesised to be the risk level. Other things being equal, workers will prefer jobs with less risk to jobs with high risk. This will result in a relative shortage of workers for risky jobs and hence wages in those jobs should be higher. This 'wage premium' then becomes a measure of risk valuation. It can be estimated by multiple regression techniques in which the wage is the dependent variable and the various factors influencing the wage are the independent variables. An example might be:

$$\text{Wage} = f(\text{Educ, Exp, Union, Risk, Occ})$$

where Educ is education, Exp is years of experience, Union is an indicator of the degree of unionisation of the labour force, Risk is the objective (or perceived) probability of fatal injury and Occ is some indicator of the desirability of the occupation. The coefficient linking Wage and Risk is then the WTA measure of risk.

One obvious problem with such approaches is that workers have to know about the differences in risks and, if they do, whether those perceptions coincide with 'objective' measures of risk such as the probability of a fatality in that industry. If there is no perception of risk, but risk exists, then the 'hedonic wage' (i.e. the wage premium) may be zero, seriously understating risk values. If there is a perception of risk but it is exaggerated compared to objective risk, then risk may be overvalued. Other problems include the potential for workers in risky jobs to be 'self-selecting', i.e. those tolerant of risk may be attracted into the industry in question. Lack of labour mobility will also mean that some workers will remain in jobs without full compensation for the risks involved.

What limited evidence there is suggests that workers actually overstate the risk of their jobs. But as Freeman (1993) points out, what matters for the hedonic wage model is the perception of differences in risks between jobs, not the absolute level of risk in a given job.

Most hedonic wage studies have been carried out in the USA and suggest that VOSLs range from \$2 million (1994 prices) to \$3.5 million.

Valuing Mortality Risks: Avertive Behaviour

Individuals spend money in trying to reduce risks, so called ‘averting behaviour’. Under certain circumstances these expenditures approximate the economist’s concept of WTP to reduce risk. The kinds of averting expenditures in question might be on smoke alarms; safety harnesses, tamper-proof drug storage containers, and so on. These kinds of expenditures can be regarded as part of what is called a ‘health production function’ in which the state of good health is ‘produced’ by various factors, including expenditures on averting ill-health. Note that some apparent averting expenditures are not valid measures of risk reduction. Thus, it is quite widely assumed that life insurance expenditures are measures of WTP to avoid risk. But insurance expenditures do not have the effect of reducing risks. Indeed, they may actually increase risks by encouraging less careful behaviour - the issue of ‘moral hazard’. As Freeman (1993) notes, life insurance essentially values the earning capacity of the insured individual to the dependants who are the ones who will gain from any insurance policy. This is not at all the same thing as the individual’s willingness to pay to reduce risks to his or her own life, which is what is required.

The health production function can be written:

H = f(Poll, Med, Avert, Other)

where Poll is the level of pollution, Med is the level of medical treatment, and Avert is the level of averting activity. ‘Other’ refers to all the other factors affecting health status: age, income, smoking behaviour, and so on. Reducing pollution will reduce the time spent being unwell, say from 4 days to 3 days. If by spending £X through avertive behaviour the same reduction in ill-health can be achieved, then £X should be the value of the reduced pollution to the individual. More formally,

WTP (Pollution Reduction) = (Reduced Time in Ill Health) x (Extra Cost of ‘Producing’ Health by Mitigating Activity).

In this way, expenditures on risk reduction can be interpreted as WTP for risk reduction. In practice, finding examples of avertive expenditures that are ‘purely’ health producing has proved difficult. Studies include seat belt use and smoke detectors and suggest VOSLs of about $0.7 million and $2.2 million.

Valuing Mortality Risk: Contingent Valuation

The contingent valuation method (CVM) requires that individuals express their preferences in response to a questionnaire. It is therefore very much akin to market research in which the researcher seeks to find out how a respondent would behave, in terms of WTP, for a modified or new good. Questionnaires take two forms: (a) open-ended or continuous approaches simply ask what someone is WTP (or WTA), and no prompting of likely values is permitted; and (b) discrete or dichotomous choice in which the value is posed and the respondent is then asked whether he or she is willing to pay that sum, yes or no. Yes/no questions that use the cost of providing some project or benefit as the sum to which the yes/no answer is sought are also known as ‘referendum’ approaches. There is now a general preference for the dichotomous choice format. The kinds of biases that may occur in CVM include:

(a) Starting point bias in the dichotomous choice format, i.e. respondents tend to produce WTP answers that tend towards the first ‘price’ put forward by the questioner. Such a bias is easily tested by seeing if the difference between the average stated WTP is statistically different to the starting point sum;

(b) Strategic bias whereby the respondent understates the true value of their preference (they ‘free ride’) in the expectation that others will state more and thus secure the good in question for everyone. This phenomenon was long thought to be inherent with ‘public goods’, such as clean air, since if the clean air is provided for any one individual it is provided for everyone (clean air is said to be ‘jointly consumed’). Of course, the misstatement of preferences may be biased the other way: someone may be so keen to see the good provided that they overstate their preferences, fearing that others will free ride. Tests for strategic bias suggest that, contrary to expectation, it may not be of major significance. Such tests may involve stratifying the sample deliberately to give some people strong incentives to free ride and others less of an incentive, and then seeing if their average WTPs diverge. Others involve indicating that unless more than a certain percentage of respondents vote for the good, it will not be supplied.

(c) Hypothetical bias: the respondent may produce answers that are purely hypothetical, i.e. if the good or policy in question is actually provided, their WTP will be less than stated in response to the questionnaire. Careful design of questionnaires can reduce the hypothetical bias problem to very low levels in WTP questionnaires. It may be more of a problem with WTA questions. Measuring the bias usually involves comparing stated preferences with actual preferences when real sums of money are involved in the CVM. Since respondents are less familiar with compensation contexts (the relevant context for WTA), their stated WTA is likely to exceed their actual WTA.

(d) Part-whole or ‘mental account’ bias: here the problem is that, while the questionnaire may focus on a specific environmental benefit, the respondent may act as if he or she is valuing environmental improvement in general. Tests for this kind of bias involve varying the quantity of the good in question, e.g. a 1% reduction in risk, a 10% reduction in risk and so on, and seeing whether WTP varies significantly as the benefit is increased. A number of studies suggest that WTP is the same and that what individuals are ‘purchasing’ is not the benefit in question but the ‘warm glow of giving’ or ‘moral satisfaction’. One response has been to ensure that questions are not asked until the respondent has been reminded that he or she has a specific budget to be allocated. The empirical evidence of part-whole bias remains mixed.

(e) Information bias: the quality of information supplied to the respondent may affect the stated WTPs. Usually, the more information is supplied about the risks in question the higher the WTP. Why this is regarded by some critics of CVM as a flaw in the method is unclear. Information should influence WTP in exactly the same way as information influences WTP in the everyday market place for ordinary goods. While there is an interesting issue of how much information should be provided, what matters most is that the same level of information be provided to all respondents.

There are other problems with the CVM but modern CVM design is capable of minimising the extent of error in stated responses. Good surveys of CVM are to be found in Pearce et al., (1994) and Bateman and Turner (1993).

Valuing Mortality Risk: The Human Capital Approach

Before the formalisation of hedonic wage, CVM and avertive behaviour approaches the most commonly used technique to value risk to human life was the human capital approach. The idea is simple: an individual is ‘worth’ to society what he or she would have produced in the remainder of their lifetime, gross of taxes since the interest is in society’s valuation of the individual. An argument did exist as to whether the earnings that are relevant should be net or gross of the individual’s own consumption. If the individual’s consumption is excluded then the value concept is simply that of how the rest of society values the individual, and that is inconsistent with the WTP approach. If the individual’s consumption is included, then at least some gesture is made towards included a value from the individual's own standpoint. But there is in fact nothing to suggest that an individual's WTP need be equal to the remaining lifetime income of that individual. The link to the WTP approach is clearly tenuous at least. The WTP approach is based on how individuals value risks to their own lives, whereas the human capital approach makes no obvious reference to that concept. Indeed, the human capital approach says little about individuals' attitudes to risk. Its one virtue is that it is thought to be very easy to calculate (Rowlatt et al., 1998).

Even if the human capital approach is accepted as a rule of thumb, however, there are problems in its estimation. First, if the individual at risk has retired from work, the human capital valuation would suggest that their VOSL is zero or even negative (if consumption is deducted). This perhaps under lines the failure of the concept in terms of its theoretical underpinnings. If human capital was a WTP concept, those out of work should have positive WTPs. Second, someone may be of median age but still not be producing marketed 'output'. A house person, for example, produces non-market output and this would have to be valued. Third, future earnings cannot simply be added up year by year to get a total since the individual will discount the future. Hence a discount rate is needed.

While it would be better if the human capital approach was avoided altogether, it is still widely used, no doubt because of the relative ease of computation. If used, the computation in question would be:

$$\text{'VOSL'} = \Sigma_{i=1,T-t}(p_{t+i}.Y_{t+i})/(1+r)^i$$

where $\Sigma_{i=1,T-t}$ denotes the sum over time from time t, the current age of the individual at risk, T is the age at which the individual ceases to work, p_{t+i} is the probability of the individual surviving from age t to age t+i, Y is income, and r is the discount rate.

The human capital approach does not produce a value of statistical life in the sense of Σ_i VORi,i above. But can it be used to estimate the value that others put on the life at risk? For close relatives, friend etc. the answer must be 'no': such people do not value risks in terms of the income forgone by the individual at risk. But what of society generally? There is a sense in which society loses the output of the individual less the consumption of that individual. But had the individual survived, the rest of society might have to produce transfer payments in the form of welfare payments, and illness costs arising from the ill-health that the individual would have suffered had they survived. Arguably, then, what the rest of society loses is i's income minus i's consumption minus i's claims on the rest of society over the expected lifetime of i without the risky event. This is perhaps the most that can be said for the human capital approach.

Willingness to Pay versus Willingness to Accept?

The valuation of statistical lives rests on the WTP or WTA principle. It is easy to see that WTP is constrained by income and/or wealth. WTA appears not to be so constrained since it is an amount in compensation for accepting a risk. Both the contingent valuation and wage-risk models have the capability to elicit WTA estimates and practice has found that WTA figures are not infinite, i.e. people do not expect extremely large payments in compensation for losses of environmental quality. But WTA does tend to exceed WTP, as noted earlier, and these differences cannot generally be explained by issues of questionnaire design. There are genuine differences between WTP and WTA. This raises the issue of which measure is correct?

The answer depends on the context, and especially on property rights, although other factors help to explain the size of the discrepancy between WTP and WTA. The matrix below explains the property rights issue. Generally, WTP is the right concept when the individual whose valuation is being sought does not have a right to the improvement being valued. WTA is the correct concept when the individual does have a right to the status quo and is being asked to forgo a benefit or accept a loss.

Valuation of a GAIN WILLINGNESS TO PAY => Property rights DO NOT rest with individual	Valuation of a LOSS WILLINGNESS TO PAY TO AVOID THE LOSS => Property rights DO NOT rest with the individual
Valuation of a GAIN WILLINGNESS TO ACCEPT COMPENSATION TO FORGO THE GAIN => Property rights DO rest with the individual	Valuation of a LOSS WILLINGNESS TO ACCEPT COMPENSATION => Property rights DO rest with the individual

The relevance of property rights arises in two contexts. The first is where there are clearly defined legal rights, and the second relates to the individual's perception of rights. Thus, we might expect individuals to value a unit loss much more highly than a unit gain if he or she believes they have some right to the existing amount of environmental quality or asset. There is indeed evidence that individuals have 'loss aversion', i.e. they regard the status quo as some kind of reference point from which gains and losses are evaluated. This view is stressed by advocates of 'prospect theory' - see Kahneman and Tversky (1979).

A second factor explaining the wide divergence sometimes found between WTP and WTA is the degree of substitutability of the thing being valued with other goods. Suppose, for example, that what is being valued is a unique environmental or material asset - the Grand Canyon or the Taj Mahal, say. Then, as there are no ready substitutes one might expect WTA to be very much higher than WTP. And this turns out to be the case: the fewer the substitutes the larger the discrepancy between WTA and WTP, as theory would predict (Hanemann, 1991). Hanemann's explanation is not comprehensive because the same WTA/WTP discrepancy exists for commonplace goods, in which case the insights from prospect theory appear to be relevant.

The relevance to life risks is of course significant if individuals feel that risks to life or health constitute an invasion of their 'rights' not to have to tolerate those risks. There is some evidence to suggest that those rights will be especially pronounced when the risks are not voluntary, in contrast with, say, occupational risks. If so, we might expect wage risk models, which are WTA estimates for voluntary risk, to reveal risk valuations that are above, but not substantially above, WTP valuations. CVM models, on the other hand, might reveal substantial WTA/WTP discrepancies if the risk in question is involuntary. The problem with the evidence is that most of the VOSL studies are either wage-risk studies or CVM studies of transport risk. Transport risks may or may not be seen as voluntary compared to, say, radiation risks from a nuclear power plant, although most risk studies appear to treat transport-related risks as involuntary. Early analysis of the voluntary/involuntary risk valuation issue was fairly inconclusive. Starr (1972) attempted a comparison of risk levels and the associated benefits and concluded that involuntary risk might be valued by a factor of ten more than voluntary risk. Reworked by Otway and Cohen (1975), Starr's ratio appears far too high and a factor of two appears more appropriate. But further analysis by Fischoff et al., (1979) reinstates the large tenfold differential between involuntary and voluntary risk values. Substantial question marks hang over these studies however not because of the risk data but because of the use of measures of benefit based on actual expenditures on the activity in question or the contribution the activity makes to an individual's income. There is some affinity here with the required benefit measure - WTP - but it is far from precise.

Overall, then, the conceptual contexts in which WTP and WTA should be used are fairly clear and relate to the presumption about property rights. In practice, determining the assignment of property rights is far less straightforward.

The Evidence on VOSL
Several reviews exist of VOSLs.

Pearce, Bann and Georgiou (1992) review the various estimates of VOSL and find the mean estimates across studies shown in Table 5.2 (updated to 1997 values). The estimates also show the ratio of WTA to WTP because of the presumption that WTA studies tend to find higher values than WTP studies. The wage risk studies are fairly consistent between the UK and USA with a suggestion that a higher (average) value exists for WTA in the USA than in the UK. On the other hand, WTP studies appear to produce higher values in the UK. These data also suggest that the WTA > WTP inequality holds for the USA but not for the UK, but there is no ready explanation for these disparate results. Note, however, that the estimates shown are unweighted averages, i.e. it assumed that all the studies reviewed are equally valid.

Table 5.2 Values of Statistical Life

UK£m (1997)	USA	UK
WTA (wage risk)	2.9-4.6	2.4-2.9
WTP (CVM, CRM)	1.2-2.2	3.3-5.3
WTP (market)	0.9-1.0	0.5-2.8
WTA/WTP (WR/CVM)	1.3-3.8	0.4-0.9
WTA/WTP (WR/mkt)	2.9-5.1	0.9-5.8

Source: Pearce et al. (1992) updated to 1997 prices

Other reviews for the USA suggest ranges of recommended VOSLs. Fisher *et al.,* (1989) recommend a range of $2-10 million; Cropper and Freeman (1991) recommend $2-6 million; Viscusi (1992) recommends $3-7 million and Miller (1989) recommends $1-4 million. An extensive review by Industrial Economics Incorporated (1993) fits a lognormal distribution to available estimates considered to be ‘reliable’ (26 in all) and takes the geometric mean (i.e. the mode) to obtain $4 million in 1993 values, or $4.5 million in 1997 values. Overall, then, VOSL estimates of around US$ 1.6-4.8 million would appear to be ‘safe’.

Use of VOSL estimates of the kind noted in Table 5.2 has come under criticism for several reasons:

(a) There is unease about the fact that health benefits based on VOSL are so dominant in cost-benefit studies;

(b) the VOSL estimates come largely from accident contexts where the mean age of the person killed is very much lower than in pollution contexts. There is therefore a feeling that older people, perhaps with an already impaired health state will not have the same valuation of risk as someone who is very much younger;

and

(c) it is, as noted above, very easy to confuse what a VOSL is actually measuring. Wrongly translated as a ‘value of life,’ the concept is easy prey for critics who do not invest in attempts to understand the analytical foundations of VOSL. Since this confusion is widespread, analysts often prefer not to use the VOSL concept at all.

Of these reasons, only the second has any intellectual basis, although the first does reflect a ‘statistical sensitivity’ issue in the sense that, if the VOSL estimates are wrong, then entire decisions may be changed.

For these good and bad reasons, then, there have been attempts to estimate not the value of the risk of fatality but the value of the life period gained by reducing the risk. This has come to be known as the ‘value of a life year’ or VOLY.

Values of Life Years (VOLYs)
The underlying rationale for valuing ‘life years’ is that many contexts in which health risks occur relate to pollution. Clearly, pollution is more likely to affect people who are most vulnerable. In a poor country this may be the very young and the very old. In a rich country, where infant mortality risks are very low, it is more likely to affect the elderly and especially those who are already at risk from their prevailing health state. Suppose, for argument’s sake, that, statistically, the reduced life expectancy of someone exposed to air pollution is six months. Then, the argument goes, what matters is the value the individual places on those six months of extended life. If the period is a few weeks or even days, then the relevant value is that ‘life period’ rather than the actual risk. This contrasts with the VOSL where a person, however old they are, is faced with a risk and they express their WTP to reduce that risk. In principle, the two values - VOSL and VOLY - should bear some relationship since the person at risk must have some idea of remaining life expectancy. Indeed, it would be extremely surprising if they did not. In expressing a WTP to reduce risk, then, they should be accounting for the remaining life period available to them.

One obvious way of approaching the problem is to see if WTP to reduce risks is functionally related to age, an issue we return to below. The surprising thing about the VOSL literature is that very little of it controls for age, so that only a few studies exist to offer a guide on how risk valuations vary with age.

Alternative approaches attempt to estimate the VOLY and, so far, two procedures have been used. The first simply takes estimates of the VOSL and converts them to values of life years; i.e. no additional information is sought. The second attempts to construct VOLYs from first principles by engaging in valuation studies that directly attempt to elicit the WTP for extended periods of life.

VOLYs derived from VOSLs

One approach to estimating the VOLY is to regard it as the annuity which when discounted over the remaining life span of the individual at risk would equal the estimate of VOSL. Thus, if the VOSL of, say, £1.5 million relates to traffic accidents where the mean age of those involved in fatal accidents is such that the average remaining life expectancy would have been 40 years, then

$$\text{VOLY} = \text{VOSL}/\text{A}$$

where $\text{A} = \text{A}(n,r) = \sum_{t=1}^{n} 1/(1+r)^{t}$ or $\text{A} = [1-(1+r)^{-n}]/r$.

and n is years of expected life remaining and r is the utility discount rate[6]. Examples are shown below for n = 40 years.

[6] The utility discount rate is the rate at which future wellbeing is discounted, not the rate at which income or consumption is discounted. The UK Treasury (1997) adopts a rate of pure utility discounting of 1.5% but little evidence exists to support this rate. Pearce and Ulph (1995) suggest a rate of 0.3%.

VOSL £m 1997 prices	VOLY at r=0.3% £	VOLY at r=1% £	VOLY at r=1.5% £
	A = 37.6	A = 32.8	A = 29.9
1.0	26,595	30,460	33,445
1.5	39,894	45,690	50,167
2.0	53,190	60,920	66,890
3.0	79,787	91,138	100,000

r is utility discount rate.

These VOLY numbers can then be used to produce a revised VOSL allowing for age. At age 60, for example, suppose life expectancy is 75 years. The VOSL(60) is then given by

$$\textit{revised}\ \text{VOSL}(60) = VOLY * \sum_{t=1}^{T-a} 1/(1+r)^t$$

where *T-a = 15* is remaining life expectancy. In the case indicated, this would be, at 1% discount rate and a 'standard' VOSL of £1 million:

revised VOSL(60) = (30,460).(13.87) = £422,480.

The result is that the age-related VOSL declines with age and this appears to accord with the intuition of some commentators (see the discussion below). The generalised formula for age related VOSL is:

$$\textit{revised}\ \text{VOSL(a)} = [\text{VOSL(n)/A}] * \sum_{t=1}^{T-a} 1/(1+r)^t \quad \text{or}$$

revised VOSL(a) = VOSL(n) * A(T-a,r)/A(T,r)

where a is the age of the individual or group at risk, T is life expectancy for that group, VOSL(a) is the age-adjusted VOSL and VOSL(n) is the 'normal' VOSL.

One advantage claimed for this approach to valuation is that it can be combined with other information on the health state of the individual at risk. This might be done via 'QALYs' -quality of life year ratings. QALYs involve weighting life expectation by quality factors that reflect individuals' own perceptions of the quality of life associated with that life expectancy. Extending a life by one year but with an associated level of pain and suffering thought to be unbearable would attract a low QALY indicator. A VOLY multiplied by this QALY would give a revised quality-adjusted VOLY (Davies and Teasdale, 1994).

While the VOLY approach may appear sound it suffers from a number of deficiencies.

First, it offers no evidence that VOSL declines with age in the manner shown. If this were to be the case, we would expect to find evidence that the WTP to reduce risks varies inversely with age. As Rowlatt et al., (1998) note, there *is* some evidence for a declining WTP as people become older, but that evidence is not at all consistent with the age profile of VOSL as dictated by the VOLY approach. Ignoring any influence from health states, the VOLY approach implies a monotonically declining VOSL with age, whereas the WTP for risk literature tends to produce inverted 'U' shapes. In essence, the age-related VOSLs derived on this approach are arbitrary: they are imposed from outside rather than being derived from any individual-based risk assessment. Maddison (1998) suggests that there are sound reasons for supposing that VOSL is proportional to the number of discounted life years remaining to an individual and that it is inversely proportional to the survival probability in the current time period. In other words, Maddison suggests that there are rationales for a declining VOSL with age, but that this will be attenuated in old people by the reduced survival probability. For the UK, he suggests that the VOSL for a 74-year-old with six months life expectancy would be 17% of the healthy 36-year-old.

Second, while the evidence on age and WTP for risk reduction is not compelling, what there is suggests a decline in WTP. Jones-Lee (1989, 1993) reports WTP for accident reductions in the UK and these are shown in Figure 6.1. For illustration, they are compared there to the implied VOSLs that would come from using the VOLY approach. Notice that the VOLY-based VOSLs do not exhibit the 'inverted U' shape found in the Jones-Lee studies and they seriously understate later age VOSLs when compared to the standard VOSL approach. Also, the VOSL ratios using the VOLY approach are invariant with the value of VOSL(n), but will change with the discount rate assumed. However, Figure 6.1 shows that the VOLY-based VOSL is largely unaffected by the choice of 1% or 2% utility discount rates. Supporting evidence for modest declines in WTP with age can be found in Maier et al., (1989), Miller and Guria (1991), Kidholm (1995), Persson et al., (1995) and Desaigues and Rabl (1995). Rowlatt et al., (1998) cite a Swedish paper - Persson and Cedervall (1991) - which found rising values of WTP with age, a result that Rowlatt et al., put down to problems in eliciting answers to questions about small risk changes, but which could be consistent with theory (see below). Johannesson and Johansson (1996) also find modestly increasing WTP with age.

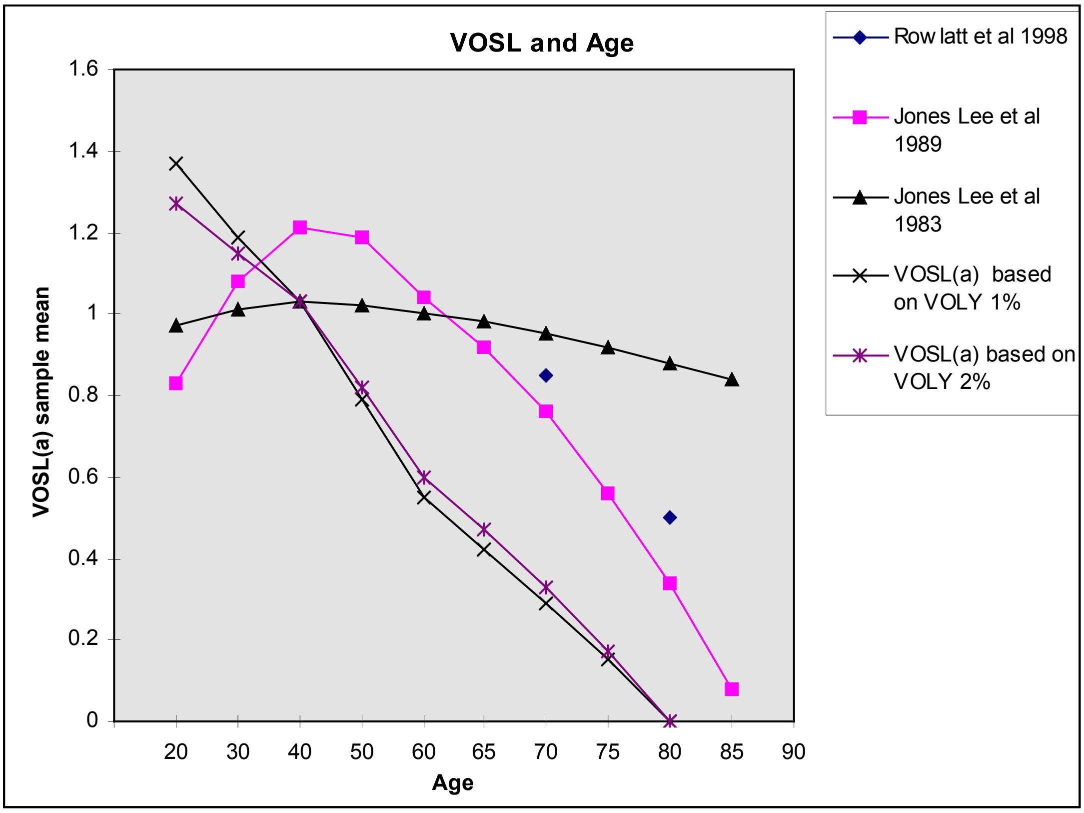

Figure 6.1 Value of Statistical Life (VOSL) as function of age according to Rawlett et al, 1998, Jones-Lee et al.,. 1989 and 1993, and for the VOLY-based VOSL approach for utility discount rates of 1% and 2% respectively.

Third, it was noted that the VOLY-based VOSL could be combined with QALY information. Again, it appears that the VOLY approach imposes an apparently 'logical form' on the valuations by assuming those already ill will value remaining life periods less. But to quote a recent study for the US Environment Protection Agency:

‘..it is possible that the reduced life expectancy and reduced enjoyment of life associated with many chronic illnesses may result in lower WTP to reduce risks of death. On the other hand, facing serious illness and reduced life expectancy may result in higher value [being] placed on protecting the remaining time.’ (Chestnut and Patterson, 1994) .

Overall, Maddison’s approach holds out some promise for finding age-related VOSLs via indirect routes. These should then be tested against VOSLs derived from direct approaches in which age is specifically accounted for.

VOLYs derived from WTP experiments

An alternative procedure based on the VOLY concept is to see the WTP to extend a lifetime conditional on having reached a certain age. Johannesson and Johansson (1996) report a contingent valuation study in Sweden where adults are asked their WTP for a new medical programme or technology that would extend expected lifetimes conditional on having reached the age of 75. Respondents are told that on reaching 75 they can expected to live for another 10 years. They are then asked their WTP to increase lifetimes by 11 years beyond 75, i.e. the ‘value’ of one extra year. The results suggest average WTP across the age groups of slightly less than 10,000 SEK using standard estimation procedures and 4,000 SEK using a more conservative approach. In dollar terms this is \$600-1500[7]. Recall that this is for one year of expected life increase. WTP actually *increases* with age, although not dramatically - on the standard basis, 8000 SEK for the 18-34 age group, 10,000 for the 35-51 age group and 11700 for the 51-69 age group. Using the formula:

$$\text{VOSL(a)} = VOLY * \sum_{t=1}^{T-a} 1/(1+r)^{t}$$

Johannesson and Johansson suggest these values are consistent with ‘normal’ VOSLs of \$30,000 to \$110,000, substantially less than the VOSLs derived previously. Since *T-a* is obviously less the older the age group, then the relevant VOSLs will decline with age. They also derive discount rates of 0.3% to 3.4% and these are invariant with age. Finally, they argue that these lower valuations are consistent with findings in Sweden and the USA on social attitudes to allocating resources to life saving. Thus, Cropper et al., (1994) found that survey respondents strongly favoured life saving programmes which save the lives of young people rather than old people. Earlier work by Johannesson and Johansson (1995a, 1995b) found that Swedish attitudes were similar, and that expectations about the future quality of life at old age play a significant role (regardless of what the actual quality of life is). The implications of the low WTP values for health care are hinted at in Johannesson and Johansson (1996): they observe that the VOSL values are ‘negligible’ compared to the costs of health treatment for the aged.

The Johannesson and Johannsson study is the only one available at present which attempts to value of life year directly. Is the WTP approach used consistent with the VOSL approach ? It is arguable that the ‘goods’ being valued are quite different: VOSL studies value risk and the VOSL is simply an aggregation of those individual valuations of risk. The WTP for a life year is not explicitly a value of risk, but a value of extending a life year once the respondent is assumed to reach a particular age. The Johannesson and Johansson paper could be argued to be more relevant for pollution control policy if the benefits of that policy are thought to accrue mainly to the elderly.

VOSL and Age Again

As opposed to accidents, environmental risks are especially likely to affect the health of those already predisposed to illness, e.g. the elderly. Hence, it is important to know if the VOSL is likely to vary with age. At one extreme we could legitimately argue that we have no reason to

[7] The range if reported as \$400-\$1500 in the original article but this looks like a misprint.

suppose this WTP will vary negatively with age. Indeed, older people may be all the more risk averse simply because the value of time itself is likely to increase the less there is of it remaining to the person at risk. Plausible reasons to suppose that WTP will fall as age increases have been advanced in the theoretical literature (e.g. Freeman, 1993, Chapter 10; Cropper and Simon, 1994). Freeman (1993) reviews life cycle models and shows that, in general, one might expect WTP to decline with age. This is because lifetime utility is dependent on lifetime consumption in such models and older people simply have fewer consumption years left. However, there are several reasons why such life cycle WTP models understate 'true' WTP:

(a) they tend to omit others' valuations of the life at risk (e.g. relatives, friends) (see below);

(b) life cycle models assume that expected lifetime utility depends on expected lifetime consumption only, whereas individuals surely value survival as well. Note that this value of survival need not vary inversely with age at all, and could actually increase;

(c) there is evidence to suggest that WTP for 'contemporaneous risk' is less than WTP for 'latent risk', i.e. WTP for avoiding accidents is less than WTP for avoiding risks of cancer (Jones-Lee *et al.,*. (1985)). Yet the empirical VOSL literature is almost entirely based on accident risks. For pollution issues, then, transferring VOSLs from accident risk contexts to pollution contexts is likely to understate the 'true' degree of risk aversion.

Others' Valuation of Risks to an Individual

The second component of the basic valuation equation was the value placed on risks to i by others who are close to i, relatives and friends. The literature that seeks to estimate such valuations is very much smaller, but suggestive of some results. Viscusi et al., (1988) surveyed consumers to elicit risk valuations for injury risks from the use of insecticides in the USA. Consumers were asked their WTP to reduce risks from 15/10,000 to 10/10,000 for two pairs of risk: inhalation and skin poisoning and inhalation and child poisoning. The WTP figures of $1.04 and $1.84 respectively, therefore implies values of risk of $2080 and $3680 (1.04/0.0005 and 1.84/0.0005). Individuals were then asked their WTP for an advertising campaign to reduce risks by the same amount generally, i.e. to other people. The results implied valuations of the first risk pair of $10,000 for North Carolina State - where the survey was conducted - and $3,070 for risks outside the state. For the second risk pair, the values were $18,100 and $4,260. The state/non-state comparisons suggest that valuations decline as the individuals at risk become more 'anonymous' to the valuer, as one might expect.

An early study by Needleman (1976) sought the valuation of close relatives for reductions in risks. The study looked at kidney donors. Donors tended at that time to be close relatives to secure greater chances of acceptance of the transplanted organ. The kidney donor suffered a slight increase in risk while the recipient had dramatically improved chances of survival. By looking at data on actual kidney donations and at refusal rates - i.e. situations in which the relatives refused to make the donation - Needleman estimated a 'coefficient of concern'. An average coefficient of 0.46 implies that close relatives' valuations may be 46% of the value of risk of the individual at risk, i.e. one might write $VOR_{i,j} = 0.46VOR_{i,i}$, where j is now close relatives. Recall that $VOR_{i,i}$ is summed across all individuals at risk and expressing a positive WTP to obtain a VOSL. It follows that $VOR_{i,j}$ should be summed across all close relatives of those at risk. The effect could be substantial. For example, if each individual at risk has four close relatives, the effect would be to multiply VOSL by 4 x 0.46 = 1.64 to obtain the summed valuations of close relatives. Schwab Christe and Soguel (1995) conduct a contingent valuation analysis of willingness to pay to avoid the consequences of a road accident. WTP was estimated in two contexts: where the respondent was the hypothetical victim and where the respondent is a relative of the hypothetical victim. In each case, the pain and suffering of others is relevant. In the former case, willingness to pay ($VOR_{i,i}$ in our notation) may already account for the pain and suffering of relatives and others, i.e. WTP is influenced by the concern the victim has for the effects of an accident to him/herself on others. In the second case, where the victim is a relative, WTP ($VOR_{i,j}$ in our notation) may reflect both the relative's own bereavement and also some judgement of the pain and suffering of the victim. Schwab Christe and Soguel try to distinguish these effects. The results are:

(a) $VOR_{i,i}$ for a death is 1.7 million Swiss francs, or around 1.2 million US$;

(b) $VOR_{i,i}$ for an accident involving severe and permanent disability is slightly higher than $VOR_{i,i}$ for death at some 1.75 m Swiss francs;

(c) $VOR_{i,j}$ for relatives (j) is *higher* than VORi,i at around 2 million Swiss francs, and higher still for permanent and severe disablement. In general VORi,j would appear to be equal to 1.25 $VOR_{i,i}$, about three times the effect found by Needleman's study.

Cropper and Sussman (1988) suggest that US citizens have a willingness to pay for children's statistical lives equal to 70-110% of their own values ($VOR_{i,i}$). This is consistent with a New Zealand study by Miller and Guriua (1992) with a $VOR_{i,j}$ of 119% for family members. Blomquist *et al.,*. (1996) estimate a VORi,i of $2 million and a $VOR_{i,j}$ for children by parents of $3-5 million, i.e. 1.5-2.5 times the $VOR_{i,i}$. Blomquist *et al.,*.(1996) also review other studies of $VOR_{i,j}$, finding a fairly consistent range of values between 23 and 50% of $VOR_{i,i}$ when the person at risk is not a family member.

The studies suggest that $VOR_{i,j}$ may be of the order of 100% for own family members and perhaps 20% for non-family members. The implications of adding 20% premia for *each person* affected by the ith life at risk are fairly significant. Not only would a typical valuation of, say, $2 million be quadrupled because of close family valuations, but a further $0.4 million (20% of $VOR_{i,i}$) might need to be added for each person thought to exhibit a degree of concern for the individual at risk. VOSLs, then, could be seriously understated by focusing on $VOR_{i,i}$ alone.

However, the issue of aggregating life risks across individuals is complex. For a discussion see Johansson (1995). Jones-Lee (1992) cautions against assuming that $VOR_{i,i}$ and $VOR_{i,j}$ can be added but suggests a social value of a statistical life of 1.1 to 1.4 times the $VOR_{i,i}$. This is based on analysis of altruistic motives. For *pure altruism* - in which the person exhibiting the concern respects the preferences of the person at risk - the correct VOSL is the 'own' valuation. The original proof is given in Bergstrom (1982). Jones-Lee (1991) examines the case of *pure paternalism* - where j exhibits a concern for i's risks but does so on the basis of overriding i's preferences - and concludes that the same result holds, i.e. $VOSL_{i,i}$ is the correct valuation. Where there is a focus by j on i's 'safety', i.e. risk reduction, and the utility function for j takes the form:

$$U_j = U(x_j, s_j; s_i)$$

where x is the private good and s is safety, then it is legitimate to add a 'premium' to the own VOSL. Thus, for any premium to be justified, j's preferences have to be paternalistic and relate only to i's safety, not to i's consumption of the private good.

Valuing Future Lives

Given that 'sustainable development' is a widely embraced goal of economic and environmental policy, and given that 'sustainability' raises the importance of impacts on future generations, one issue of some importance in risk valuation is that of how to value 'future lives'. Essentially, should a life at risk in, say, 50 years time be valued in the same way as a life today ? This is an 'intergenerational equity' issue. Jones-Lee and Loomes (1993) have shown that, on balance, future lives should be valued at the current VOSL and should *not* be discounted. Or, put another way, the effective discount rate applied to future lives should be zero provided the valuations being applied are the current VOSLs. In benefit-cost analysis a similar result would be obtained by valuing future lives at a future VOSL, i.e. one allowing for the expected growth of incomes which will therefore make future generations more willing to pay for risk reductions, and then discounting that value to get back to a current value. So, for a life risk 50 years hence we would have two alternative rules for valuation at the current period:

$$VOSL_{t=50} = VOSL_0,$$

the ‘equal values no discounting’ rule

or $$VOSL_{t=50} = VOSL_0.e^{50g}.e^{-50r}$$

the ‘discounted future values’ rule, where g = expected rate of income growth and r = the discount rate. So long as r=g the two rules are the same. The rules become more complex once we allow for the degree of aversion to inequality that might be displayed by the current generation; once a distinction is made between the discount factor for future risks and the discount factor for future income; and once survival probabilities vary between generations. In general:

(a) the greater the degree of aversion to inequality, the closer one gets to the equal values and no discounting case;

(b) the greater the survival probabilities of future generations relative to current generations the more justified is discounting future risk reduction benefits; and

(c) only if future wellbeing (as opposed to income) is discounted, can discount rates greater than zero be justified in the context where the current VOSL is used to value future risks.

More generally, either future risks are valued at future WTP levels and then discounted in the same way as income, or future risks are valued at current VOSLs and no discounting is allowed, provided there is impartiality between current and future generations.

Valuing Statistical Lives When Incomes are Unequal

WTP and, less obviously, WTA estimates of VOSL are constrained by income. WTP and WTA estimates are also averages, i.e. there is a frequency distribution from which the mean is taken, so that some people have much higher valuations of risk than the mean and some have much lower valuations than the mean. One of the reasons for these different valuations will be income differences within the nation. This procedure for deriving a VOSL has given rise to extensive misunderstanding. Imagine two countries, one rich and one poor, such that the rich country imposes a risk on the poor country through pollution. Global warming, which results from the emission of greenhouse gases, is often regarded as an example of such ‘imposed’ pollution costs. (Although the rich world (the OECD countries) actually emits just under 40% of total greenhouse gases, with 60% coming from the developing world, oil rich nations, and the ex-Soviet Union (World Resources Institute, 1994)). Estimates of VOSLs determined by WTP estimates in the rich and poor countries will produce higher values for the rich country than the poor one, WTP being (partly) determined by income levels. Suppose the rich country’s pollution gives rise to an estimated 100 premature mortalities in the poor country. Assume the rich country faces the choice of spending resources on international pollution control to the benefit of the poor countries, or spending the same level of resources on a domestic issue which also saves 100 lives, i.e. the marginal cost of saving lives is the same in the two countries. A cost-benefit test will result in the resources being spent domestically because the 100 ‘domestic’ lives will be ‘worth’ more than the 100 overseas lives due to the higher risk valuations. Yet if ‘all lives are equal’ in some sense, such an outcome seems very unfair, especially if the rich country can be said to impose the pollution on the poor country. Some have argued that, if a VOSL is to be used at all, it should be the same VOSL for everyone and that VOSL should be the higher of the two figures, i.e. the VOSL for the rich country.

Is the benefit-cost test then invalid in some moral sense? There are several issues to be distinguished.

First, the VOSL within a country is an average, as noted above. In principle, then, the same procedure should be used where VOSLs differ across countries. The resulting VOSL will be an average of the two VOSLs, but it will not be the highest VOSL that is used. If the highest figure was chosen, then, logically, it must also be chosen within a country, i.e. the average should not be used. Such an outcome is not logically tenable since the individual with the highest aversion to risk would then determine everyone's valuations. The idea of averaging valuations to reflect concern about the inequality of WTP is a long standing one in cost-benefit analysis. Pearce (1986, original edition 1971) discusses a rule in which WTP is weighted by a ratio of average income to actual income, i.e. an adjusted WTP for any country i becomes:

WTPi* = WTPi.□ /Yi

and WTPj is

WTPj* = WTPj.□/Yj

where □ is the average of Yi and Yj. The ratio of the two WTPs is then

$$WTPi^*/WTPj^* = \frac{WTPi.Yj}{WTPj.Yi}$$

This procedure will produce the same 'common VOSL value' if the value of risk as a proportion of income is the same in both i and j. Only if WTPj as a proportion of Yj is higher than WTPi/Yi will the resulting VOSL be higher in j than in i, and if the proportion is higher in i than in j, then the weighted VOSL will be higher in i than in j.

Second, regardless of the equity weighting procedure discussed above, the cost benefit test need not produce the unfair outcome discussed in the example above. This is because the example assumes a common marginal cost of reducing risks in both countries. In practice, risk reduction is likely to be less costly in the poorer country than in the rich country. Even with 'unequal lives' then, nothing follows about the outcome of a benefit-cost test.

Conclusions

Few topics have proved so controversial as the 'value of statistical life'. In large part the controversy derives from unfortunate terminology, since what appears to be at stake is the 'value of life' itself. This confusion has not been helped by even the most distinguished commentators and analysts using this phrase. But what is being estimated is the value of risk reduction. VOSLs are, essentially, convenient ways of aggregating these estimates.

In a finite world there really should be no dispute that resources have to be allocated rationally across different life risks. The real focus of the debate should be on the size of the VOSL. As we saw, this is the subject of a debate, which centres on two approaches to valuing risks. The first asks for the WTP to avoid risks, and the second asks for the WTP to extend an expected lifetime by some finite period, say one year. The literature on 'value of life years' turns out to be a hybrid of these approaches, deriving VOLYs from a given VOSL. As discussed, there appears to be limited theoretical justification for this hybrid approach. It is also not consistent with what we know about VOSLs as they vary with age. Nonetheless, what we know about the age-WTP relationship is not much. In turn, the literature that attempts directly to estimate VOLY is minute.

Such as it is, it suggests VOSLs are very much less than those derived from standardised VOR calculations.

Other issues concern the role that others' valuation of risks should play and the role that discounting might play in valuing future risks. In general it would appear that there is a case for adding a modest premium to own VOSLs for others' paternalistic concerns, and there is no strong case for discounting future risks.

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