

**VALUING RISKS TO LIFE AND HEALTH
Towards Consistent Transfer Estimates
in the European Union and Accession States**

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**Paper prepared for the European Commission (DGXI)
Workshop on Valuing Mortality and Valuing Morbidity,
November 13, 2000, Brussels
Revised, December 2000**

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1 The issue

Health benefits, in the form of reduced premature mortality and reduced morbidity, figure prominently in cost-benefit studies of actual and proposed European Directives on environmental quality control. Table 1 shows a selection of studies relating to air pollutants and reveals that health benefits account for a minimum of one-third and a maximum of nearly 100 per cent of overall benefits from pollution control. Moreover, in most cases these benefits exceed the costs of control by considerable margins. Health benefits therefore 'drive' positive benefit-cost results¹. Nor is this outcome peculiar to the European Union. The US EPA's retrospective and prospective assessments of the Clean Air Act produce extremely high benefit-cost ratios, e.g. 44 for the central estimate of benefits and costs (US EPA, 1997). Moreover, EPA regards these as probable underestimates. In turn, the benefits are dominated by health benefits (99% if damage to children's IQ is included). The EPA's analysis has, however, been subjected to very critical analysis (Lutter, 1998; Sieg et al., 2000). By contrast, the European studies appear not to have attracted much by way of critical comment².

It may be the case that there are very high benefit-cost ratios for air pollution control, but there are at least two reasons for a feeling of unease about the results that are being obtained.

- (1) The relevant studies tend to omit ecosystem benefits, despite the fact that, for acidifying substances in the wider Europe, ecosystem protection is the driver for the UN ECE region air pollution Protocols under the Convention on Long Range Transport of Air Pollution (LRTAP). If the presumption of the Convention Parties that ecosystem damage is of dominant importance is correct, this would suggest that benefit cost ratios are substantially higher than the factors of three to five being recorded in the European studies. Some would regard this as adding to doubts about the analysis, rather than reducing them.
- (2) The European studies suggest that benefits exceed costs even for scenarios defined in terms of 'maximum technologically feasible reduction' (MFR) of pollutants, i.e. scenarios in which the most pollutant-reducing technologies are used. Such scenarios should be characterised by very high marginal abatement costs at very high levels of pollution reduction, precisely the context where one would expect incremental benefits to be less than incremental costs. While the benefit cost ratio does appear to fall for such scenarios relative to other more modest abatement targets, the reduction is not dramatic and benefits continue to exceed costs. Thus, AEA Technology (1999) finds a benefit cost ratio of 2.17 for a MFR scenario, compared to 2.87 for practical targets based on the relevant Protocol. The incremental benefit cost ratio of going from Protocol targets to 'MFR' targets is 1.6.

What then accounts for the high value of health-related benefits? The underlying equation in all of the studies is simply stated as:

¹ It is noteworthy that many of the studies listed in Table 1 have been carried out by one agency, AEA Technology. In turn, AEA Technology uses mainly the 'ExternE' unit values for VOSL and morbidity (CEC, 1995, 1998). In theory, the resulting high benefit cost ratios and the dominance of health benefits could therefore be the result of 'author bias', but this seems very unlikely. See text for discussion.

² Some criticisms of the original ExternE work, on which AEA Technology's valuations are based, are to be found in Maddison (1999).

Table 1 Health benefits as a percentage of overall benefits in recent cost-benefit studies

Study	Title and subject area	Benefits as % total benefits
Holland and Krewitt, 1996	<i>Benefits of an Acidification Strategy for the European Union: reductions of SO_x, NO_x, NH₃ in the European Union</i>	86-94%. Total benefits cover health, crops and materials.
AEA Technology, 1998a	<i>Cost Benefit Analysis of Proposals Under the UNECE Multi-Effect Protocol: reductions of SO_x, NO_x, NH₃, VOCs</i>	80-93%. Total benefits cover health, crops, buildings, forests, ecosystems, visibility
AEA Technology, 1998b; Krewitt et al, 1999.	<i>Economic Evaluation of the Control of Acidification and Ground Level Ozone: reductions of NO_x and VOCs. SO₂ and NH₄ held constant.</i>	52-85% depending on inclusion or not of chronic health benefits. Total benefits include health, crops, materials and visibility
AEA Technology, 1998c	<i>Economic Evaluation of Air Quality targets for CO and Benzene</i>	B/C ratio of 0.32 to 0.46 for CO. Costs greatly exceed benefits for benzene. Benefits consist of health only.
AEA Technology, 1998d	<i>Economic Evaluation of Proposals for Emission Ceilings for Atmospheric Pollutants</i>	B/C ratios of 3.6 to 5.9. Health benefits dominate.
AEA Technology, 1999	<i>Cost Benefit Analysis for the Protocol to Abate Acidification, Eutrophication and Ground level Ozone in Europe</i>	VOSL + morbidity accounts for 94% of benefits. B/C ratio = 2.9.
IVM, NLUA and IIASA, 1997; Olsthoorn et al, 1999.	<i>Economic Evaluation of Air Quality for Sulphur Dioxide, Nitrogen Dioxide, Fine and Suspended Particulate Matter and Lead: reductions of these pollutants</i>	32-98%. Total benefits include health and materials damage

Note to Table 1: we have selected results using VOSL (value of statistical life) rather than 'VOLY' (value of a life year) since the latter are not correctly estimated in the studies that also provide VOLY results. See text for discussion.

$$H_{ij} = b_{ij} \cdot V_j \cdot P \quad \dots [1]$$

where

H_{ij} = the health effect, j , from pollutant i , aggregated across the relevant population

b_{ij} = the dose-response coefficient relating pollutant i to effect j .

V_j = the willingness to pay (accept) to avoid (tolerate) the health effect.

P = population at risk.

If H is 'large' it follows that one or more of b_{ij} , V_j or P is large. The main focus in this paper is on b and V , i.e. on the epidemiology of pollution or risks and V , the unit values applied to health effects³. In particular, we wish to know if we have reliable estimates of V_j that can be used for purposes of benefits transfer in the European Union and Accession countries. It turns out that the reliability of V_j estimates is not always independent of what we believe about b_{ij} as well. Hence we cannot avoid some discussion of dose-response coefficients.

2 Benefits transfer and the European Commission questions

The Commission wishes to know if it is valid to adopt a 'common' set of monetary values for health effects in EU-15 and Accession countries, or if there is a default value that can be varied according to geographical context. This is a question about the validity of the 'transfer method' or 'benefits transfer' (BT). BT is defined as 'the use of existing information designed for one specific context to address policy questions in another context' (Desvousges et al, 1998). Terminology varies, but we use the 'study site' to refer to the original context in which willingness to pay estimates have already been derived using 'primary' research (revealed or stated preference studies) and the 'policy site' to refer to the context to which the transferred value is made.

The Commission's original questions are set out below:

- can we use one value for mortality and morbidity regardless of context?
- how do we account for social and income differences within populations?
- what values should be used for Accession States
- should we use value of statistical life or value of life years?
- should values be adjusted for quality of expected life?
- how should future impacts be valued?
- what are the main sources of uncertainty?
- what research is going on? what research still needs to be done?

A suggested re-organisation of these questions is as follows:

³ But the value of P is also important. It is not always clear over what population risks should be aggregated. For example, definitions of rural and urban populations can be ambiguous, see Pearce and Crowards (1996).

Q1 *Should we transfer unit values (V_j) or benefit functions ($V_j = V_j(X_1, \dots, X_n)$) where X_1, \dots, X_n are the determinants of V_j ?*

Q2 *If we transfer functions, what is the minimum set of explanatory variables we should account for? We define the X_1, \dots, X_n as contextual variables⁴. They cover:*

- i) characteristics of the individuals at risk or whom are affected, including individuals' perceptions of the harm.
- ii) the nature and source of the risk to health
- iii) the institutional context.

Q3 *What are we trying to value?* The object of value may be a risk at a defined moment of time; a risk over a defined lapse of time, including a remaining lifetime; a risk weighted by some factor to reflect the quality of life over some period of time. This is mainly the issue of values of statistical life (VOSLs) versus values of life years (VOLYs).

Q4 *What are the main sources of uncertainty ?*

Q5 *How do we value future health risks?* Here we might define two elements of future risk:

- i) valuing risks to different individuals in the future
- ii) valuing risks to the same individual in the future

Q6 *What research should be done, if any?*

One issue that is perhaps implicit in the Commission's original list of questions is whether it is 'fair' to adopt different values for the same health effect in different geographical contexts. Fairly obviously, since the determinants X_1, \dots, X_n will unquestionably vary by location, so will the values of V_j . In a quasi-federal context such as the European Union, or a federal context such as the USA, it might appear unfair to adopt different values for different States. Otherwise, an investment in, say, life-saving might appear worthwhile in States with high values of 'statistical life' and not worthwhile in countries with low values of statistical life. Hence we have a further question:

Q7 *What role should be played by considerations of geographical equity in determining transferable values?*

3 Transferring unit values versus transferring benefit functions

In benefits transfer, it is possible to (i) transfer an average willingness to pay (WTP) estimate from one primary study, (ii) transfer mean WTP estimates from meta-analyses of existing primary studies, and (iii) transfer a WTP function.

The most elementary procedure is to 'borrow' an estimate of WTP. The estimate may be left unadjusted, or it may be adjusted in some way. Transferring unadjusted estimates is clearly hazardous, although it is widely practised. Reasons for differences in average WTP include:

⁴ We use the term 'context' to cover all the factors likely to affect the value of V_j . In the BT literature, context is often used to describe only one set of factors, namely the source of the risk (air pollution, water pollution etc.).

- Differences in the socio-economic characteristics of the relevant populations;
- Differences in the physical characteristics of the study and policy site;
- Difference in the proposed change in provision between the sites of the good to be valued, and
- Differences in the market conditions applying to the sites (for example variation in the availability of substitutes).

As a general rule, there is little evidence that the conditions for accepting unadjusted value transfer hold in practice. Effectively, those conditions amount to saying that the various factors listed above all hold, i.e. sites must be 'identical' in all these characteristics. An alternative is therefore to adjust the WTP estimates in some way. Once it is accepted that there is little if any validity in transferring unadjusted values, it must also be accepted that WTP will vary by location. It is worth noting that the studies reported in Table 1 take simple averages of VOSLs from primary studies. Several potential errors are therefore embedded in the resulting benefit estimates:

- i) the averages are taken from mixes of US and European studies, not just European studies
- ii) the averages are not adjusted for features of the studies, i.e. no meta-analysis is performed on the primary studies
- iii) the same VOSL is used for all EU and all EIT countries, i.e. no account is taken of income or other differences. This may reflect an 'equity' judgement, e.g. that EITs should be treated no differently to EU-15 populations, but, if so it is unclear why a Europe-wide average is not used rather than an EU-15 average, even if the latter does include USA values as well.
- iv) There is an additional question of whether the source-context of most of the VOSL studies (accidents) is transferable to other source-contexts, e.g. pollution, radiation hazards, chemical risks etc. This is a major issue and is discussed later.

A widely used formula for adjusted transfer is:

$$V_{ijp} = V_{ijs} (Y_p/Y_s)^\epsilon \quad \dots[2]$$

Where i is the i th pollutant, j is the health effect, p is the policy site, s is the study site, Y is income per capita, V is willingness to pay, and ' ϵ ' is the 'income elasticity of WTP', i.e. an estimate of how the WTP for the environmental attribute in question varies with changes in income⁵. We define $(Y_p/Y_s)^\epsilon$ as the 'transfer multiplier'.

In the case of equation [2], the feature that is changed between the two sites is income only, perhaps because it is thought that this is the most important factor resulting in changes in WTP. But it should also be possible to make a similar adjustment for, say, changes in age structure between the two sites, changes in prior health states, and so on. Making multiple changes of this kind amounts to transferring *benefit functions* (see below).

⁵ ϵ is not to be confused with the income elasticity of *demand*, η . The relationship is $\epsilon = \eta \cdot \partial WTP / \partial Y$, where Q is quantity. A value of $\epsilon < 1$ is consistent with $\eta > 1$, i.e. with risk reduction being a 'luxury' good. There appear to be no a priori restrictions in the value of ϵ .

The least researched element of equation [2] is the value of ϵ . This value is crucial when making adjusted transfers of this kind from, say, EU-15 to Accession States. For example, the ratio of per capita income between the Czech Republic and the UK is 0.23 (based on World Bank data). The relevant transfer multiplier for different values of ϵ would then be:

For $\epsilon = 1$ multiplier = 0.23
 For $\epsilon = 0.5$ multiplier = 0.48
 For $\epsilon = 0.3$ multiplier = 0.64

Clearly, the value of ϵ matters so long as income differences are significant⁶. Table 2 assembles what little is known about this value. Table 2 shows values of ϵ for environmental benefits generally as well as for health risks specifically, on the grounds that risk values will be embedded in some of the environmental values.

Table 2 Estimates of the elasticity of WTP with respect to income		
Study	Context	Value of ϵ
<u>Environment</u>		
Pearce (1980)	Survey of available literature	'Inconclusive' but no evidence $\epsilon > 1$
Kriström and Riera (1986)	Survey of available literature	$\epsilon = 0.2$ to 0.3 , and always < 1
<u>Risk</u>		
Day (1999)	Meta-analysis of WTA literature	Linear model $\epsilon = 2.65-3.56$ Log model $\epsilon = 0.36-0.55^*$
Krupnick et al (1996)	Interpretation of US CVM literature	$\epsilon = 0.35$
Loehman et al (1979)	WTP for morbidity reduction, USA	$\epsilon = 0.26$ to 0.60

Notes to Table 2: * 0.55 is preferred by the author

Table 2 suggests a provisional conclusion that values of $\epsilon = 0.3$ to 0.6 could be used for transfer purposes. But it is also clear that the basis for this recommendation is not a strong one and that there is a need for more rigorous research into the income elasticity of WTP for changes in risk and health states.

The use of a 'central' value for ϵ does not resolve other issue relating to transfer. First, we need to know if the base value being transferred, i.e. WTP_s , should come from a single study or multiple studies. The consensus of the literature is that transferring single estimates is extremely hazardous (Brouwer, 1998). This affects many cost-benefit studies where single studies, or a few studies only, are used to derive transfer estimates.

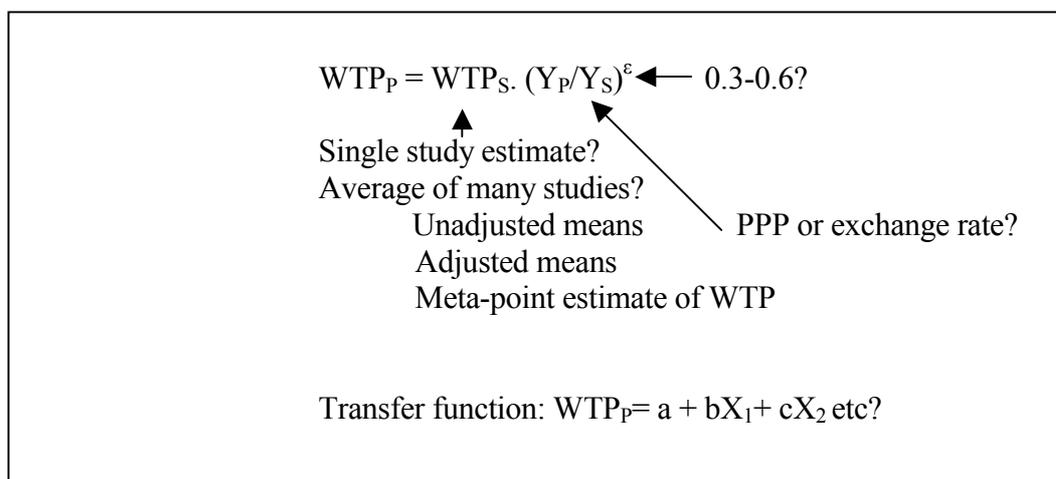
⁶ If the income ratio is, say, 0.8, the resulting multipliers are close to unity, e.g. 0.89 and 0.93 in the cases shown.

If multiple estimates of WTP are to be used, the issue then arises as to whether some simple average of the estimates is used, or whether average values might be weighted in some way -e.g. by the dispersion about the mean (the wider the dispersion the lower the weight the estimate would receive)-, or whether what is transferred is a central value based on fuller meta-analysis. Again, the literature on BT seems clear that transferring 'meta-estimates' of means is better than transferring simple averages (Brouwer, 1998).

Finally, the issue arises as to whether or not it is better to transfer *functions* rather than best estimates, however derived. Transferring a function involve estimating a meta-equation from available study sites such that the determinants of WTP are elicited and the relevant coefficients relating to them to WTP are estimated. The function - i.e. the functional form, the independent variables and the coefficients - itself is then transferred, and the relevant values of the independent variables are substituted into it from the policy site. Brouwer and Spanincks (1999) find that transferring functions is more robust than transferring averages, however derived, but also observe that function transfer can still involve very large errors. Kirchoff et al. (1997) reach similar conclusions. The options are summarised in Figure 1.

Brouwer and Spanincks (1999) argue that the explanatory power of WTP equations is often low, and that 'other factors', besides the usual factors of income and site characteristics, may be accounting for the failure of transferred WTP functions to explain WTP at the policy site. They suggest that 'attitudinal' and 'cultural' factors may be important. Whatever it is that improves the explanatory power of transferred functions, it is clear from the literature that averages, however derived, cannot be transferred without far more detailed scrutiny of the validity of making such transfers. This involves *validating* the transfers by conducting a BT exercise and primary studies simultaneously. Ideally, the transferred value and the primary estimate should be similar.

Figure 1 Options for transferring WTP (WTA) estimates



If this exercise is repeated until a significant sample of studies exists in which primary and transferred values are calculated for policy sites, then there would be a justification for assuming that transferred values could be used in the future without the need to validate them with primary studies. A more sophisticated approach takes a set of n primary studies and uses n-1 of the studies to estimate the value at the nth site. That 'transferred' value can then be compared with the original primary value at that site.

Certain conditions probably have to be met for a valid transfer of value to take place. These are:

- the studies included in the analysis must themselves be sound;
- the studies should contain WTP regressions, i.e. regressions showing how WTP varies with explanatory variables;
- the study and policy sites must be similar in terms of population and population characteristics. Alternatively, differences in population must be accounted for;
- the change in the provision of the good being valued at the two sites should be similar;
- site characteristics should be the same, or differences should be accounted for, and
- property rights should be the same across the sites.

Some general findings from the literature are:

- Transferring benefit functions is more accurate than transferring average values;
- Contingent valuation studies appear to perform no worse than revealed preference studies in terms of transfer error;
- but transfer error using stated preference studies is generally quite large, 1-75% if 'outliers' are ignored, but up to 450% if they are included;
- There is some reason to suppose that individuals' attitudes are important determinants of WTP in stated preference studies, yet most BT makes little effort to test for variability in attitudes across sites. This suggests that BT would have to be supplemented by social surveys at the policy site.
- Meta-analysis of contingent valuation studies can explain a reasonable proportion of the variation in the original studies, but the original studies do not include sufficient information to test whether more information would have increased the explanatory power of the meta-analysis, and
- The missing information may well be of the motivational type, i.e. why people adopt the value stances they do.

Overall, and as a general proposition, BT cannot, at the moment, be relied upon to produce valuation estimates which are statistically indistinguishable from the 'true' values. It may be that values are transferable but that much more information is required before meta-analyses can explain the variation in WTP across studies. On this view, more research will improve the reliability of BT at some stage in the future. At the moment, there is no consensus on these issues. This points the way towards (a) continued reliance on primary studies, and (b) conducting those primary studies in a manner that is consistent with future BT tests since many primary studies are currently unsuited to meta-analysis. As noted later, a possible exception is some forms of morbidity where there may be adequate information for valid transfer.

4 VOSL meta-analyses

The value of a statistical life (VOSL) is given by the equation:

$$VOSL = \frac{\sum_i WTP_i \cdot \Delta r_i}{\sum_i \Delta r_i} \quad \dots[3]$$

Where $\sum WTP_i$ = sum of individual WTPs (WTAs) for the change in risk over N individuals

Δr = the change in risk

N = number of persons exposed to the risk

$\sum_i \Delta r_i$ = number of statistical lives gained (lost) = $N \cdot \Delta r$

It should be noted in passing that the smaller is Δr , the higher will VOSL be for any given WTP. The relevance of this observation is that many policy contexts involve small changes in risks. It follows that even modest WTPs for such small changes can easily result in a substantial figure for VOSL

While there is a large number of studies on VOSL, there appear to be only three attempts at meta-analysis: (Desvousges et al, 1998; Day, 1999; and van den Bergh et al. 1997). All seek to estimate an adjusted mean WTA for risk increments. There have also been studies which have sought to fit a distribution to available WTP/WTA estimates. Thus, IEI (1993) suggest a geometric mean of \$3.6 million (1993 prices) for the VOSL based on a lognormal fit to the distribution of VOSLs in various studies. This is an example of fairly simple averaging.

Table 3 summarises the results of the meta-studies. All studies use WTA based on wage risk studies. The limited number of non-US studies means that US estimates have to be included in the meta-sample, possibly presenting problems for potential transfer of values to the EU. Only the study by Day (1999) includes more than one EU study, and this influences the resulting adjusted mean WTA estimate because of the recent literature in UK labour market studies suggesting very high compensation for risk levels. The study by van den Bergh (1997) focuses mainly on statistical features of the studies and offers far fewer insights into BT potential than the others. The Desvousges et al (1998) study is designed in such a way that VOSLs are transferable to low risk contexts, but is also designed so that marginal WTA does not vary with risk levels. Probably the most informative study is that of Day (1999). The principal findings of this study are:

- Around half the variation in WTA in the original studies is explained
- Risk variation in occupations is likely to be small, so the preferred explanatory regression excludes risk
- WTA varies with income, with a preferred income elasticity of WTP of 0.55⁷
- WTA varies positively with unionisation of the labour force
- Male workers exhibit a higher WTA than female workers
- WTA for risk is higher in the UK than in North America, but the UK studies tend to endogenise risk whereas the North American studies do not. Failure to account for the fact that risky jobs may be occupied by less risk-averse people means that the resulting VOSLs are not representative of a more risk-averse population. Put another way, 'true' VOSL will be under-estimated by failure to endogenise risks (but there are some doubts about the procedures used to endogenise risks in some of the relevant studies).
- Use of US Bureau of Labour Statistics results in an upwards bias in the WTA due to the fact that BLS data tend to under-estimate risk.
- WTA may be overstated because of a failure to account for non-fatal risks, i.e. actual WTA reflects not just risks of death but risks of non-fatal injury as well.

What can be learned from the meta-studies of VOSL as to the legitimacy of transferring values?

⁷ Risk and income are, however, collinear in the model.

First, all three meta-studies are almost entirely based on wage-risk studies. This appears to be unavoidable since most VOSL studies take this form. But the resulting magnitude is a WTA not a WTP, and the relationship between WTA and WTP measures is strongly disputed in the literature. Arguably, WTA estimates exceed WTP estimates. Without explicit tests for this effect in a meta-analysis, the most that can be done is to compare unadjusted means of WTA and WTP. Pearce et al (1992) suggest that studies available, at that time, $WTA < WTP$ in the UK but $WTA > WTP$ in the USA. An alternative view is to argue that, while wage-risk studies measure WTA, WTA is equal to WTP at the margin since those who avoid marginally riskier jobs are expressing their WTP for the added safety of doing so. Perhaps more of an issue is the validity of hedonic wage risk studies in a valuation context. It is well known that 'ideal' procedures for estimating WTP (or WTA) in hedonic studies involve a two-stage process. The first stage derives the partial derivative of the hedonic wage function with respect to risk. The second involves estimating the marginal willingness to pay function, the implicit price being just one point on those functions (Freeman, 1993). But this two stage procedure is rarely carried out, so that one may question whether the hedonic wage-risk estimates are measuring a 'true' WTA or WTP. Clearly, far more research is needed to illuminate these relationships.

Second, occupational risks do not capture the vast variety of risks faced by the general public. As such, it is very unclear if WTA estimates from wage-risk studies can be applied without contextual adjustment, even where the context is risk of death. As will be discussed below, the nature of the risk and institutions will be relevant.

Third, the WTA studies relate to risks perceived by working adults rather than risks faced by older people who may well have left the labour force. This raises the issue of the relationship between age and WTP which is discussed in more detail below.

Fourth, the risks faced in labour markets tend to be immediate. Transferability to contexts where risks are 'latent', i.e. realised by individuals at a later point in their life, is therefore open to serious question.

Fifth, Day's meta-analysis suggests that wage-risk studies may be measuring endogenous risk, whereas the risks of relevance to public policy will tend to be exogenous. Indeed, it is unclear if endogenous risks should be the subject of policy at all since they can be argued to be internalised in the relevant market, in this case the labour market.

Sixth, the available meta-studies are not helpful in terms of how WTP/WTA varies with the *level of risk*. Yet a major issue is how one transfers from contexts where risks are identifiably 'high', as in occupational risk, to contexts where risks may be very low, as with air pollution for example. Most studies appear to find little relationship between risk levels and WTP, an

Study	Original studies	VOSL	Comment
Van den Bergh et al. 1997 (a) OLS regressions	10 US and 1 UK wage-risk studies	\$3.86 million ('most reliable estimate')	Significant explanatory variables include sample size, no. of risk variables, t-statistic on risk > 1.8
(b) Rough set theory			Only sample size matters
Desvousges et al, 1998	28 wage-risk and 1 CVM study, USA	Constant (w.r.t risk) VOSL of \$3.6 m, with confidence interval \$0.4 to 6.8 million.	Dependent variable is WTA = VOSL/r. WTA regressed on risk and data sources.
Day, 1999	16 wage-risk studies = 10 USA, 2 Canada, 4 UK.	Best estimate of \$5.63 million	See text

example of 'insensitivity to scope' (Jones-Lee et al. 1985; Smith and Desvousges, 1987; Hammit and Graham, 1998). What one concludes from insensitivity to scope is open to question, however. One view is that the results of such studies fail to obey a basic validity requirement, namely that WTP should vary with quantity. Others might argue that such insensitivity is simply a 'fact of life'.

While meta-analysis is the proper research context to determine the basic potential for transferability, available meta-studies have been able to test for only some of the relevant factors explaining variations in VOSL. Notable omissions include the different levels of risk likely to be encountered by the general public, and different kinds of risk, e.g. risks of cancer, risks of large-scale accidents etc. Put another way, full contextual analysis of VOSL is still missing. The available studies are not therefore very helpful for transfer purposes.

5 Morbidity meta-analyses

To our knowledge, only one full meta-analysis of morbidity valuation estimates exists. This is the extensive study on morbidity in the European Union (plus Norway) carried out by Pearce et al (1999). The ExternE estimates, for example, are heavily dependent upon a very few studies from the USA. The Pearce et al. study carried out contingent valuation surveys in Portugal, the Netherlands, Norway, Spain and the UK for health effects that were thought to be associated with air pollution. An explicit effort was made to test for the effects of *context* by eliciting values for health end points without any reference to context, and repeating the exercise for the same endpoints but with some contextual material added to the questionnaire. Finally, the validity of BT was tested by estimating what the value would be in any one 'policy' country based on the values derived in the other (study site) countries, and then comparing this BT estimate with the actual value derived from the contingent valuation study in the 'policy' country. The resulting error was defined as:

$$E = [WTP_T - WTP_{CV}] / WTP_{CV} \quad \dots[4]$$

where

WTP_T = transferred WTP

WTP_{CV} = original CVM estimate.

Table 4 provides the central estimates.

Table 4 Values for morbidity in Europe: Euros WTP to avoid an episode						
	Pooled	Neths	Norway	Portugal	Spain	UK
Hospital	490	453	482	480	682	262
Casualty	253	205	382	296	234	210
Bed	155	114	190	141	181	133
Cough	43	45	58	45	62	32
Eyes	56	64	50	112	85	22
Stomach	56	-	-	98	-	42

Source: Pearce et al (1999). UK£ converted to € at 1.6:1

The relevant categories in Table 4 are: hospital = hospital admission for the treatment of respiratory disease; casualty = emergency room visit for relief from respiratory illness; bed = 3 days spent in bed with respiratory illness; stomach = one day of persistent nausea or headache; cough = one day with persistent cough; eyes = one day with itchy, watering eyes. The categories relate to respiratory illness because this was the context of the study. However, the valuations were designed to be 'context free' in the sense that the causes of the illness were not identified. Further analysis showed that the introduction of 'context' made no statistical difference to the estimates of WTP. In principle, then, these WTPs could be transferred from one location to another regardless of context since they are context-free values and context is arguably not an influence. The reliability of such a transfer exercise would partly rest on whether all contexts are accounted for. The study tested for context in the contingent valuation surveys by adopting different questionnaires: one in which context was absent, and one in which the causal context was cited. In the UK survey a further contextual dimension was added, namely the policy context, i.e. a description of policies that would reduce air pollution. By and large, 'causal' context does not affect WTP, although the Portuguese survey found a lower WTP when context was cited. The more detailed UK survey also found that policy did influence WTP with WTP being significantly higher in the 'with policy' case. As will be discussed shortly, context takes on many different aspects.

The final issue is the error, E , involved in the transfer exercise. The average value of E was 0.36, i.e. there would be an average 36% error involved in transferring estimates to a country outside the five countries studied. Most probably, this is an acceptable degree of error in cost-benefit studies and could easily be incorporated in sensitivity analysis. For the within-sample, errors were as small as 2% for 'hospital' in Norway (i.e. taking the WTP for hospital from the other four countries and applying it to Norway) but as high as 111% for the UK for hospital, and 235% for 'eyes'. As a general proposition, transferring estimates to the UK appears particularly error-prone (a range of 23% to 235%).

The transferability of the morbidity WTP estimates thus appears fairly safe in principle, provided the values sought are context free. In the more likely case (of relevance to the Commission, anyway) where WTP for proposed policy changes is sought, one might add a premium to the context-free estimates, but far more research would be needed to establish what this premium (or discount) is according to different policy contexts.

Table 4 is of interest for other reasons. Reading across the rows, the WTP to avoid states of illness is not correlated with income, e.g. the WTP to avoid illness is highest, or second highest, in Spain for hospital, bed, cough and eyes. This probably reflects the different forms of health care available in the different countries. For the within-country studies, WTP was found to be correlated with income and WTP was *positively* associated with age (see the comparable discussion about age and VOSL below).

For interest, Table 5 compares the EU study estimates with those of ExternE, recalling that the ExternE estimates were taken from the then available literature (i.e. excluding the Pearce et al. study) and were specifically used for BT purposes.

Table 5 Comparison of morbidity values in Pearce et al (1999) and those in ExternE and Maddison (2000): Euros per episode

	Pooled values from Pearce et al (1999)	ExternE values	Maddison (2000)
Hospital	490	7870	na
Casualty	253	223	na
Bed	155	75	195
Cough	43	7.5	72
Eyes	56	7.5	61
Stomach	56	75	121?

Notes: categories are not identical in the studies. Hospital and casualty are the same. A respiratory bed day is taken to be the same as restricted activity day in ExternE but the bed-day may be more restricted. Cough and eyes are minor restricted activity days and correspond to the ExternE minor restricted days. Stomach is a day of work lost and does not have a direct counterpart in the ExternE study, so it is taken here to be a restricted activity day. 'Stomach' is also assumed to be equivalent to Maddison's restricted activity day. All of Maddison's values relate to an episode of one day's duration.

The ExternE values work reasonably well for casualty and stomach, but bed, cough and eyes all appear to be underestimated in the ExternE procedure.

Table 5 also shows a separate set of estimates, taken from Maddison (2000a)⁸. Maddison's estimates are relevant because they are derived from a form of meta analysis in which an overall transfer function is derived. Maddison follows the analysis of Reed Johnson (1996; see also Desvousges et al, 1998) by integrating 'quality of wellbeing' (QWB) indexes with WTP estimates. QWB estimates are cardinal indicators of wellbeing based on a 0 to 1 scale, death to

⁸ We have adjusted the Maddison estimates for inflation and converted to Euros at 1.6:1. Maddison's estimates cover a wider range of health effects but not casualty and hospital. Other effects are major asthmatic attack (€169), lower respiratory infection (€72), respiratory symptoms (€72), acute bronchitis (€169), chest discomfort (€96), minor RAD (€72), phlegm (€42).

perfect health. Maddison adds some Norwegian data to the US data used in Reed Johnson and derives the following meta-equation:

$$\ln WTP = 1.76 - 4.80 \cdot \ln QWB + 0.49 \ln DAYS \quad \dots [5]$$

where DAYS is the duration of the illness. Note that as QWB falls, WTP increases sharply. This could be seen as being consistent with the original European WTP estimates in Column 1 of Table 4.

While there are few estimates to compare, Maddison's results do not seem far removed from those derived from the original CVM studies reported in Table 4. This suggests that further meta-analysis in which the WTP data are taken from the CVMs conducted in the five European countries could be used to test the QWB-WTP approach further, rather than relying on predominantly US studies.

6 Morbidity: cancers

There may be a higher WTP to avoid cancers than other diseases. This is because of the 'dread' effect of such a serious illness. Fatal cancers could be valued at the relevant VOSL (see above), although in a context where there is prior knowledge of the likely cause of death, the 'dread' factor could increase WTP. Non-fatal cancers (NFCs) may attract values that are unique to those illnesses. It is somewhat surprising that the valuation literature has comparatively little to say about the values attached to cancers.

Rowe *et al.* (1995) adopt a value based on the US costs of treating cancers ('cost of illness', COI) and then multiply this by 1.5 on the basis that, where COI and WTP studies are available, WTP appears to be 1.5 times the COI. This procedure is clearly not satisfactory, as there are few studies that estimate COI and WTP. Moreover, the Rowe *et al.* COI value dates from the mid-1970s. Their valuation is some 6% of the VOSL they use.

ExternE uses a figure of \$450,000 for an NFC (i.e., some 15% of the VOSL), but it is unclear how this sum has been derived.

Viscusi (1995) conducts a computer experiment in which respondents are able to trade off ill health against risk of death in an automobile accident. His results (for the USA) suggest that a curable lymph cancer would be valued at some 63% of the VOSL, which, in this case, would give a value of about four times the suggested ExternE figure.

Murdoch and Thayer (1990) estimate WTP for skin cancer avoidance using a 'defensive expenditures' approach—i.e., by looking at changes in expenditure on sun protection products. They find that the total damages from anticipated increases in non-melanoma cancers are about one-half of the COI measure used by the US Environmental Protection Agency at the time. In undiscounted form, their estimates can be shown to result in a value per case of around \$30,000.⁹ However, most of the cases occur well into the future. Values for skin cancer are clearly not comparable to those for pollution or radiation-induced cancers, since the vast majority of skin cancers are operable with only slight effects.

⁹ Estimated by taking their estimated 2.96m extra cases and an undiscounted defensive expenditure of \$87.7 billion.

Aimola (1998) uses the contingent valuation method (CVM) to elicit cancer risk valuations from a small sample of the population in Sicily. The cancers in question were prostate, uterus, leukaemia and lung cancers. The results of these studies are summarised in Table 6.

Table 6 Economic valuation of NFCs (Euros 1999)

Rowe et al., 1995	USA	186,000	NFCs generally, based on COI
ExternE	Europe	450,000	Source unknown
Viscusi, 1995	USA	1,950,000	Lymph cancers
Murdoch and Thayer, 1990	USA	30,000	Skin cancers
Bryant, 1992	Australia	16,000	Skin cancers, COI;
		7,000–150,000	Skin cancers, CVM
Aimola, 1998	Italy (Sicily)	50,000	Lung cancer
		90,000	Uterine cancer
		500,000	Prostate cancer
		730,000	Leukaemia

7 The influence of context on WTP

The analysis in Section 5 suggested that the causal context of health end points might not matter in terms of influencing WTP, but, speculatively, the policy context, i.e. the way in which the problem is hypothetically solved, may matter. However, context is far wider than these two factors and it is worth reviewing all the factors that make up context. As far as possible, we comment on the likely directions of influence that each contextual factor may take.

7.1 Classifying context

Reliable transfers require either that the policy site is similar in all characteristics to the study site, or that dissimilarities can be controlled for. The relevant characteristics relate to the good that is being valued (amenity, risk reduction etc.), the factors explaining the source of the risk and the policy context for its reduction, and characteristics of the population whose values count.

7.2 Nature and quantity of the good

Section 5 discussed the classification of end-points for morbidity. It is essential that a primary value for any health endpoint, H_j , is transferred to an identical or near-identical health end point in the policy site. Thus, a cough is cough, but a cough of three days duration is not the same as a cough of one day's duration. Nor can it be assumed that any value V_j is proportionate, a cough of three days' duration may be valued more than three times the value of a cough of one day's duration, and so on. As far as morbidity is concerned, the available evidence is not sufficient to provide guidelines on what the appropriate adjustments should be for reliable transfer. This is a

situation that can only be improved by substantially more investment in good BT validation tests and more primary valuation study.

With respect to mortality risks, a death would seem to be a death and all deaths should, at first sight, be valued equally. But we have already seen that people appear to be insensitive to the scope of risk, i.e. higher risks may not attract higher WTP. As noted earlier, this could be interpreted as a refutation of the validity of transfers simply because a scope test is not met and hence the axioms of expected utility are violated. Alternatively, it could simply be that people are insensitive to the scale of risk, in which case it may be legitimate to transfer WTP estimates based on one level of risk to a policy context with another level of risk. Evidence for the former conclusion comes from studies that find people are often incapable of understanding risk factors, especially in contexts where risks are small (Jones Lee et al, 1985; Hammitt and Graham, 1998).

But scale of risk expressed as probabilities of death does not capture the full nature of the risk. 'dread' factors may be important: the value of mortality risk where the risk occurs as a cancer may be entirely different to the value associated with a sudden accident. Deaths can, in this sense, be *acute* (occurring immediately), *chronic* (a 'lingering' death), or *latent* (occurring at a later period due to a risk encountered now). A priori, WTP for different types of death is likely vary¹⁰.

The literature that explores WTP for different contexts of the good being provided is comparatively modest in size. McDaniels et al. (1992) found that WTP to reduce 'familiar' hazards (e.g. road accidents) was mainly determined by the perception of the individual about the extent to which they were personally exposed to the risk. WTP for less familiar hazards was influenced by dread factors and the perception of the severity of the accident. Mendeloff and Kaplan (1990) found that WTP varied according to the kind of accident giving rise to death: occupational exposure, cancer from chemical exposure and child deaths. Savage (1993) found that WTP to avoid stomach cancer was substantially higher than WTP to avoid road accidents, air accidents and deaths from domestic fires. Other factors can also be important. Individuals perceive some risks as being unavoidable, others as being under their control (voluntary risks) and others as being imposed upon them (involuntary). The voluntary-involuntary distinction is linked to feelings of responsibility. Risks for which the individual holds themselves responsible will tend to be valued lower than risks for which the individuals hold others responsible. Risks of death on the London Underground may attract a 50% premium over fatality risks on the road, due mainly to the involuntary nature of the Underground risks. Domestic fires appear to attract a 25% discount relative to road fatalities due to feelings of personal responsibility (Rowlatt et al, 1998). Rowlatt et al. (1998) suggest multipliers for air pollution fatality risks of 1.75-2.35 relative to road accident risks, but these reflect a professional working hypothesis rather than a research finding.

Voluntariness is further linked to controllability. In principle, risks undertaken voluntarily should be controllable through avertive behaviour. In turn avertive behaviour should reduce risk aversion since risks become endogenous rather than exogenous. Hence, the more feasible avertive behaviour is, the lower the implied VOSL is likely to be. Römer et al.(1998) test for risk endogeneity in a study of hazardous waste site risks in Berlin. They find that inclusion of

¹⁰ In classificatory terms, issues such as dread reduction can be regarded as a feature of the good in question, or as a feature of individuals. We have allocated it to the nature of the good here.

averting behaviour variables (e.g. avoiding contaminated water and food supplies) substantially increases the explanatory power of the equation explaining WTP. WTP is significantly reduced for averting behaviour on drinking water, as it is for individuals who participated in complaining to the authorities about contamination. The implications of this study, if they could be generalised, would be that WTP for an involuntary or uncontrollable risk cannot be transferred to a context where the risk is controllable, and vice versa. Again, the difference between controllability and internalisation also needs to be explored. If controllable risks equal internalised risks then those risks are not relevant to public policy.

A final and much debated issue, particularly in the context of risks of nuclear power accidents, is whether WTP varies with the 'scale' of the risk. Scale here refers to the number of persons who may be at risk in a single risk event. It has been widely suggested that individuals experience 'disaster aversion' such that, say, the death of 10 individuals in one event would be valued at more than 10 times the WTP to avoid 10 deaths occurring as one in each of ten accidents. Using a questionnaire approach, Jones-Lee and Loomes (1994) report no evidence for disaster-aversion where the latter is defined in terms of 'group deaths' of 30 individuals on the London Underground. They suggest that the absence of disaster aversion may reflect the non-controllability of such risks.

Others have suggested working 'rules of thumb' for valuing group deaths. If p is the probability of an accident and N is the number of persons affected, then 'popular' disaster aversion formulae are

$$\begin{aligned} D &= pN^2 \\ D &= pN^{3/2} \\ D &= 300pN \end{aligned} \quad \dots[6]$$

where D is damage or deaths (which can then be multiplied by the appropriate money value). Note that these functions can be compared to the expected value of damage which would be pN . Each disaster aversion function can then be described in terms of its ratio to the expected value approach to derive an implied *risk premium* (RP):

Example, 1000 deaths

$D = pN^2 \Rightarrow RP = N$	RP = 1000
$D = pN^{3/2} \Rightarrow RP = N^{1/2}$	RP = 32
$D = 300pN \Rightarrow RP = 300$	RP = 300

There are two ways of interpreting such functions: (a) as real descriptors of how individuals perceive group accidents and (b) as prudent rules of thumb. As noted, Jones-Lee and Loomes (1995) find no evidence of such aversion. However, comparatively few attempts have been made to measure risk aversion for large scale group death. Slovic et al. (1979) found that perceptions of risk from nuclear power, for example, compounded numerous factors: fear of the unknown, distrust of science, exaggerated ideas about the consequences of an accident, lack of control (involuntary nature of the accident), the invisible nature of the risk etc. There was evidence that respondents regarded 'group deaths' as being more significant than individual deaths. Rules of the 'fN' type could be thought of as being consistent with the 'precautionary principle' which is embodied in European treaties of Union. Nonetheless, it seems clear that, at the moment at least, there is little empirical basis for such rules of thumb.

It has been suggested that expected utility approaches will provide theoretical support for group deaths being valued more highly. Table 7 presents some findings of the ex ante, expected utility approach expressed as a ratio of expected utility (EU) to expected value (EV) estimates in the context of nuclear accidents.

Table 7 Some suggested ratios of expected utility to expected value for nuclear risks

Study	Type of risk	Result: ratio of EU to EV
Krupnick et al. 1993	US nuclear accident = 6.2×10^{-5}	1.04 - 2.69 for 'plausible' parameter values. Could be up to a factor of 78 for damage = to 50% of income
Eeckhoudt et al, 1997	Hypothetical European accident, low probabilities	20
	Similar to Krupnick et al.	1.04-2.44
Ascari and Bernasconi, 1997	As above but adjusted for rank dependent probability (RDP)*: 10 ⁻⁵ probability 10 ⁻⁶ probability	141- 202 660-1430
	As above with 'disappointment aversion' *	4- 8.5

Notes: rank dependent probability is a procedure for placing a higher weight on low probabilities than on high probabilities, in keeping with the psychometric literature. Disappointment aversion weights significant losses highly, the weights being low if outcomes are better than expected and high if worse than expected.

It can be seen that use of the expected utility approach implies that the expected value damage estimates should be multiplied by factors of perhaps 2 or 3, and possibly 20. The major changes, however, come from the variant of the ex ante approach in which the probabilities themselves are changed. Here the risk premium (the ratio of EU to EV) is of an order of magnitude similar to that obtained by the 'rule of thumb' precautionary approaches.

The validity of multiplying WTP by disaster aversion factors remains questionable. Plausible theory can be developed to support such a procedure but empirical studies for 'large' accidents have not been undertaken such that multipliers could be developed. In the meantime, available rigorous research suggests that such multipliers may not exceed unity at least for 'modest' group deaths. In other respects, it is clear that the nature of the risk reduction and the institutional context do matter for WTP, so that no blanket rules for transferability can be derived in the absence of further information.

7.3 Characteristics of the population

The second major category of contextual effects relates to the characteristics of the population whose values 'count' at the policy site. We ignore perceptions of risk since these have been dealt with in Section 6.2 under characteristics of the good being provided. The other relevant characteristics are age, income, health status, 'culture', and the population over whom values are to count -i.e. who has 'standing'. The age issue also raises the problem of what exactly is being valued: a risk or 'time remaining'?

Age

Most VOSL estimates are derived from road accident and occupational risk contexts. Here average age may be around 40 years. In contrast, other risks affect different age groups. A major nuclear accident, for example, could affect the entire age distribution of a given population, whilst other radiation risks may be confined to young children. Air pollution is more likely to affect older people and, perhaps because of correlation, persons who are already ill. Unless the VOSL is invariant with age, WTP cannot be transferred from one site to another without adjustment for age structure. Oddly, it is only recently that efforts have been made to integrate age as an explanatory variable in WTP for risk reductions.

What evidence there is on age and WTP for risk reduction suggests a probable decline in WTP as age increases. Jones-Lee (1989) found a pronounced 'inverted U' shaped curve, i.e. low WTP at low age, then a rising WTP, then a lower WTP at old age. Jones-Lee et al. (1993) reported a flatter curve, i.e. with no significant effect of age on WTP. More recent research by Jones-Lee and colleagues finds the inverted 'U' shape again (UK Department of Health, 1999). 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). Krupnick et al (2000) found that age does not affect WTP until a threshold of around 70 years of age.

Contrary views come from several sources. Persson and Cedervall (1991) found rising values of WTP with age, a result, however, 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, although this study has been severely criticised (Krupnick et al, 1999)¹¹. The European contingent valuations for *morbidity* in Pearce et al (1999) found WTP varying *positively* with age. As far as mortality is concerned, then, only criticised studies appear to find WTP increasing with age. The morbidity studies have yet to be published.

Table 8 reports the suggested adjustment factors for age taken from the Jones-Lee et al. recent work, and from Krupnick et al (2000)¹². The table also shows earlier Jones-Lee et al. studies for reference, with the results being shown in italics. However, selection of the two recent papers should not be taken as conclusive proof that VOSL falls with age. The theoretical justification for expecting VOSL to fall rests with the lifetime consumption model. But, to quote one study for the US Environment Protection Agency:

¹¹ Krupnick et al (1999) point out that it was a telephone survey, the nature of the good was not well defined and the risk change was large.

¹² See also Krupnick et al. 1999 for a pilot CV study in Tokyo where respondents are asked to value risk changes occurring over the next ten years and risk changes from age 70-80.

'..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).

Table 8 Possible ratios of age-specific VOSL to mean VOSL by age group from two studies

Age group	Jones-Lee et al. (1989)	Jones-Lee et al. (1993)	Jones-Lee et al. (Dept of Health 1999) UK	Krupnick et al (2000) Canada
40	1.21	1.03	1.00	1.00
45			1.00	1.00
50	1.19	1.02	1.00	1.13
55			1.00	1.13
60	1.04	1.00	1.00	1.13
65	0.92	0.98	1.00	1.13
70	0.76	0.95	0.80	0.72
75	0.56	0.92	0.65	0.72
80	0.34	0.88	0.50	0.72
85	0.08	0.84	0.35	0.72

Table 8 suggests that age does not affect WTP until age 70, a result that is reasonably consistent across the two recent studies. But the ratios for the over 70s are very different, with the UK study suggesting half of the relevant ratio value for Canada.

Overall, there is better evidence from recent studies that WTP falls with age, but only after age 70. The exact nature of the multiplier remains indeterminate until further studies test for the age-WTP relationship. In the meantime, a working hypothesis would be that those older age groups at risk from air pollution have WTPs of a third to three quarters of the 'mean' WTP.

What is being valued?

One issue in the VOSL literature is exactly what it is that should be valued in risk valuation studies. If someone aged, say, 75, expresses a WTP for a risk reduction, it is reasonable to suppose that the relevant risk poses an immediate threat. This suggests that the correct valuation concept is WTP for the risk reduction at that time. In terms of life expectancy, the good being valued is the avoidance of a reduction in expected lifetime. If the person expressing the valuation is aged 40 and the risk is immediate, then, again, the relevant concept is the WTP for that risk reduction. But if the person expressing the risk is aged 40 and the risk is latent, i.e. it will be realised at some point in older age, then the relevant WTP appears to be what that person is WTP today for a risk reduction in the future, against the baseline of what life expectancy would otherwise be. This final concept invokes the issue of discounting which is intimately involved in the 'value of life years' approach - see below.

Whether the question is asked in terms of risk reduction or time survived, the benefit is fairly immediate as long as the age group in question consists of those groups most at risk. For

someone of median age, however, the question could be about immediate risks (accidents) or about latent risks (pollution).

Does it matter which question is asked? Even at the intuitive level, WTP to reduce a contemporaneous risk -i.e. a risk immediately affecting the respondent - should be translatable into a WTP for lifetime. But it is also easy to see that there could be differences, such that what is being valued in each case is not quite the same. In the contemporaneous case, what is being valued is the probability of not reaching the expected age at which life will end anyway. In the lifetime case what is being valued could be an extension to the expected age at which life would otherwise end (the 'value of longevity'). The issue is whether questionnaire approaches give consistent answers to two separate questions, one based on WTP for risk reduction, the other based on WTP for life a period of life. In theory, respondents should be able to see the relationship between a question about risk and its implications for remaining lifetime. In practice, it is open to question as to what they believe about this linkage. This may not be surprising when it is recognised that those asking the questions may themselves be unclear what the link is.

Thus, one of the surprising features of the epidemiological literature is that there is little evidence of just what this expected gain in lifetime actually is from risk reductions. Maddison (1998) applies the analysis of Pope et al. (1995) to the UK for an hypothetical wholesale elimination of particulate matter, and estimates that the change in the conditional life expectancy of the 80+ age group is 1.1 months; that for the 70-79 age group is 2.1 months, and the 60-69 age group is 3.0 months. Maddison nonetheless secures very high aggregate values for particulate air pollution in the UK using a value of a life-year (VOLY) approach, the high values for the older the age groups reflecting the higher risks of mortality from air pollution for those groups, and hence higher WTP, and the influence of the discount rate (which has little effect on older groups' WTP but a big discounting effect on younger groups). The point here is that the concentration of risks, such as those associated with air pollution, among the older-aged groups in society, might appear to suggest that the relevant aggregate social value should in turn be very low due to the short periods of life that are saved by reducing those risks. In other words, VOSLs of even one third of the baseline VOSL might seem too high if those risks are translated into life expectancy lost. This doubt seems to underline repeated criticisms of using VOSL to value air pollution risks: how can large benefits accrue nationally from air pollution reduction when only short periods of life are saved? Maddison's 1998 work suggests that, even when lifetime saved is estimated in terms of months, the resulting economic values can still be very large (indeed, in his case, larger than those estimated by Pearce and Crowards (1996) for the UK using a standard VOSL approach). The reason is that, while lifetime saved may be small, the WTP for those savings among older age groups is high because the risk to them is high and the risk is contemporaneous. It was noted earlier that an undiscounted WTP is the relevant valuation for older age groups.

But in a more recent study of air pollution and mortality in Manchester in the UK, Maddison (2000b) reaches what appear to be rather dramatic epidemiological results as far as acute pollution is concerned. He notes that previous studies have generally introduced pollution variables into mortality models either as a contemporaneous risk or with one or two lags. Impacts are transient and epidemiologists resist implying anything about the length of time by which life is shortened because of acute episodes of pollution. Maddison uses rational lag procedures to estimate the time period over which acute effects occur. He finds that particulate matter ('Black Smoke') and NO₂ are significant at the 1% level of significance, O₃ at 5% and

SO₂ is not significant. Of greater importance, he estimates the change in the mortality risk ratio arising from an increased 'dose' of air pollution. Much of the effect occurs on the first day. On the fourth day following the pulse of pollution there is a marked *reduction* in the number of deaths. Essentially, those who would have died then have been 'harvested' earlier by the immediate effect of air pollution. But these counterbalancing effects fade away after the fourth day. By the fifth day, the mortality risk ratio has returned to 'equilibrium'. The basic implication is that we cannot reject the hypothesis that lives are foreshortened by only four days.

If it is the case that lives are shortened in *acute* cases (the analysis naturally says nothing about chronic effects) by only a matter of days, the issue arises of whether this coincides with the perceptions of those who might state a WTP to avoid the risks associated with acute air pollution. In other words, would 'high' WTP reflect an understanding that the issue is one of days of life rather than months or years, or would it reflect a misperception about the link between risk and expected lifetime? Perhaps what it suggests is that stated preference approaches need to be much clearer about what it is that respondents are being asked to value, and what those respondents believe about the links between risk reduction and lifetime saved.

Health status

It seems to be thought generally that the health status of those at risk also affects WTP. If so, transferred WTP values should account for any differences in health status between the study and policy sites. This may be relevant to transfers from the EU to the Accession states. For example, life expectancy in EITs tends to be lower than in EU countries: Czech Republic relative to Finland is -3 years, Poland - 4 years, Estonia -8 years, Lithuania -7 years, Russian Federation -10 years (UNDP, 2000).

UK Department of Health (1999) conjectures that the lower is life expectancy, the lower will be WTP. This effect is regarded as being additional to the adjustment for age. For example, VOSL for someone aged 75 would be 65% of the baseline VOSL (see table 7 above). Suppose life expectancy for this group is 10 years. Now consider someone within this group who, due to ill-health, has a life expectancy of one year only. Then, one adjustment would be to compute the relevant WTP as:

$$0.65 \text{ VOSL}(\text{baseline}) \times 0.1 = 0.065 \text{ VOSL}(\text{baseline}).$$

Those with only one month's life expectancy would have a value of $0.065/12 = 0.0054$ (baseline VOSL), and so on. Whilst interesting as rules of thumb, these adjustments are no more than that. They do not reflect any empirical research.

Finally, it is thought that WTP may vary directly with the state of health of the person at risk. Those in low states of health would be expected to have lower WTPs than those in better states of health. There is epidemiological evidence that those at risk from pollution-induced mortality are already in relatively poor states of health. The evidence suggesting that those in poor states of health have lower WTPs is however not conclusive. Johannesson and Johansson (1997) suggest there is a positive relationship, but Krupnick et al (2000) find no relationship between health status and WTP. Indeed, Krupnick et al (2000) found some evidence that having cancer could *raise* WTP.

The evidence linking health status to WTP is scanty and inconclusive. It certainly does not support the view that WTP should be lowered for those in poor health.

Culture and attitudinal variables

It was noted earlier than, in the context of environmental benefits, culture and attitudinal variables may play a role in improving the explanatory power of WTP equations. We have no evidence on this as far as health states and mortality risks are concerned beyond the evidence relating to attitudes to risks already discussed.

Gender

We noted earlier that few studies test for gender differences in WTP. Krupnick et al. (2000) suggest that males may have a lower WTP for risk reductions, but gender was not statistically significant in their regressions. Day's (1999) meta analysis of wage risk models suggests that males have a higher WTA than females. Clearly, further work is required, but the effect does not seem to be a priority issue for research.

Whose values?

While most of the focus of the risk valuation literature has been on 'own' valuations, there is a literature that asks about the aggregation of valuations by others for one individual's life risks. 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 $WTP_{m,n} = 0.46WTP_{m,m}$, where m the individual at risk and n is a close relative. $WTP_{i,j}$ 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 $4 \times 0.46 = 1.64$ 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 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 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) $WTP_{m,m}$ for a death is 1.7 million Swiss francs, or around 1.2 million US\$;
- (b) $WTP_{m,m}$ for an accident involving severe and permanent disability is slightly higher than $WTP_{m,m}$ for death at some 1.75 m Swiss francs;
- (c) $WTP_{m,n}$ for relatives is *higher* than $WTP_{m,m}$ at around 2 million Swiss francs, and higher still for permanent and severe disablement. In general $WTP_{m,n}$ would appear to be equal to 1.25. $WTP_{m,m}$.

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. This is consistent with a New Zealand study by Miller and Guria (1992) with a $WTP_{m,n}$ of 119% for family members. Blomquist *et al.*(1996) estimate a $WTP_{m,m}$ of \$2 million and a $WTP_{m,n}$ for children by parents of \$3-5 million, i.e. 1.5-2.5 times the own valuation. Blomquist *et al.* (1996) also review other studies of $WTP_{m,n}$ finding a fairly consistent range of values between 23 and 50% of own WTP when the person at risk is not a family member.

Overall, the, the studies suggest that $WTP_{m,n}$ 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 own WTP 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 $WTP_{m,m}$ and $WTP_{m,n}$ can be added but suggests a social value of a statistical life of 1.1 to 1.4 times the $WTP_{m,m}$. 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. $VOSL_{m,m}$ 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 health

Valuing 'future lives' is obviously of importance in the context of 'intergenerational equity' issues such as those posed by threats such as global warming, nuclear spent fuel and waste storage, toxic dumps etc. 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:

$$\text{VOSL}_{t=50} = \text{VOSL}_0,$$

the 'equal values no discounting' rule

or
$$\text{VOSL}_{t=50} = \text{VOSL}_0 \cdot e^{50g} \cdot e^{-50r}$$

the 'discounted future values' rule, where g is the expected rate of income growth and r is 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.

8 VOSL or VOLY?

Section 6 raised the issue of what exactly it is that is being valued through the techniques used to derive VOSLs. The underlying rationale for valuing 'life years' is that many contexts in which health risks occur relate to pollution. Which 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 1 months.

Then, the argument goes, what matters is the value the individual places on that 1 month 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 at risk, however old they are, is faced with a risk and they express their WTP to reduce that risk. As noted earlier, the two values - VOSL and VOLY - should bear some relationship since the person at risk must have some idea of remaining life expectancy. In expressing a WTP to reduce risk, then, they should be accounting for the remaining life period available to them. We noted that how far actual questionnaires imply this linkage is open to question.

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

$$\text{VOLY} = \text{VOSL}/A \quad \dots[7]$$

where $A = [1-(1+r)^{-n}]/r$

and n is years of expected life remaining and r is the utility discount rate¹³. Examples are shown below in Table 9 for n = 40 years¹⁴.

Table 9 Deriving VOLYs from VOSLs: examples			
VOSL €m	Utility discount rate = 0.3%. A = 37.6	Utility discount rate = 1.0%. A = 32.8	Utility discount rate = 1.5%. 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

These VOLY numbers can then be used to produce a revised VOSL allowing for age. At age 60, for example, suppose life expectancy is 15 years. The VOSL(60) is then given by

$$\text{VOSL}(60) = \Sigma \text{VOLY}/(1+r)^{T-60}$$

where T is life expectancy. In the case indicated, this would be, at a 1% discount rate and a 'standard' VOSL of €1 million:

$$\text{VOSL}(60) = (30,460).(13.87) = \text{€}422,480.$$

¹³ 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 (1999) suggest a rate of 0.3%. Some of the literature appears to use very high utility discount rates, of 5% and above. It is easy to see that such rates could be correct for the overall (social) discount rate, but hard to see that they can be construed as utility discount rates.

¹⁴ Another way of saying the same thing is that $\text{VOLY} = \text{VOSL}/\text{Discounted expected lifetime}$. Strictly, the relationship holds only when utility of consumption is constant in each time period.

The result is that the age-related VOSL declines with age and this appears to accord with the findings noted earlier that WTP probably does decline with age. The generalised formula for age related VOSL is:

$$VOSL(a) = [VOSL(n) / A] \sum_t 1/(1+r)^{T-a} \quad \dots[8]$$

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 a 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.

Are VOLYs derived from a VOSL legitimate?

There are several reasons for doubting the usefulness of the VOLY approach when it is based on a VOSL.

First, the basis of the VOLY approach is the life-cycle consumption model with uncertain lifetime (see, for example, Freeman, 1993, ch.10). It is well known that such models assume utility depends on consumption alone and not on the length of life. Lifetime utility does indeed vary with life expectancy but the route is via consumption not via time itself. It seems unlikely that individuals are indifferent to time remaining. There are also additional restrictions on the model to ensure that WTP is proportional to the discounted value of life expectancy. Thus, it can be questioned whether the underlying theory needed to derive VOLYs from VOSLs is itself tenable. (For an elegant analysis in which the life-cycle model is analysed under both expected utility and rank-dependent utility, and in which QALYs can be integrated, see Bleichrodt and Quiggin, 1999).

Second, the theory forces the age-distribution of VOLYs to take on a monotonically declining form: VOLY simply declines with age. As noted earlier, it seems more likely that actual WTP follows an inverted 'U' shape curve. If so, the VOLY construct is a poor representation of 'true' WTP over the lifetime of individuals.

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. Note that this is a different 'good' to the one implicit in VOSL studies. What is being valued is an extension to expected lifetimes, not the reduction of a risk of not achieving the expected lifetime. Johannesson and Johannesson (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 expect 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 \$600-1500¹⁵. 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:

$$VOSL(a) = VOLY \sum_t 1/(1+r)^{T-a}$$

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.

In the pollution context, WTP to avoid risks X years hence would be relevant only for the minority of younger people making up the population at risk. As noted previously, what should dominate the aggregate WTP for pollution risk reductions should be the WTP of older persons for what is, to them a contemporaneous risk.

9 Cost of illness

While most of the research on values of mortality and morbidity focus on 'own' WTP to avoid risks or to secure risk reductions, there is a continuing interest in the notion of 'cost of illness' (COI). COI seeks to measure potential losses to the economy as a whole from morbidity. Two categories of cost have been researched:

- (a) the cost of lost output due to absenteeism arising from morbidity induced by factors such as pollution;
- (b) the cost to national health care systems of treating illness.

Lost production

If pollution induces morbidity which in turn affects workplace output, the 'lost' output represents a real cost to the economy. Who precisely bears that cost will depend on how labour markets function. If absenteeism is rationally forecast by employers, wage rates or salaries should be lower than they otherwise would be in the absence of absenteeism due to this cause. In this case, employees bear the costs: wages are simply lower than they otherwise would be. If some form of

¹⁵ The range is reported as \$400-\$1500 in the original article but this looks like a misprint.

legislation insulates employees from bearing these costs, for example minimum wage legislation, then employers bear the costs in the form of reduced profits. The extent to which they bear these costs will in turn depend on their market power to shift cost increases forwards.

Pearce et al. (1999) present benchmark estimates of *per diem* costs that would arise from production losses, based on reported wage rates. These are shown in Table 10.

Table 10 EU Per diem production losses due to absenteeism. €2000			
Country	Cost	Country	Cost
Belgium	-	Netherlands	67
Germany	75	Austria	-
Greece	20	Portugal	25
Spain	50	Finland	-
France	-	Sweden	59
Ireland	53	UK	58
Italy	59		
Luxembourg	132	EU average	60

Source: adapted and updated from Pearce et al, 1999.

As would be expected, costs vary directly with the different economic circumstances of the individual countries.

What is the evidence linking pollution and absenteeism? Hansen and Selte (2000) report significant statistical links between PM₁₀ and absenteeism in Oslo, but far weaker links between other pollutants and absenteeism. A rise of 1µg/m³ in PM₁₀ was found to induce a 0.6% increase in absenteeism due to sick-leaves. Zuidema and Nentjes (1997) similarly found associations in the Netherlands, although results were sensitive to the model used.

Health care costs

Pearce et al. (1999) also investigate health care costs in the EU. These are shown in Table 11 and include both capital and operating costs.

Table 11 Health service costs €2000		
Country	Emergency room visit	Cost per hospital in-patient day
Belgium	-	315
France	33	433
Germany	28	417
Italy	23	-
Netherlands	-	508
Spain	132	-
United Kingdom	94	329

Source: Pearce et al (1999) all converted from original data to 2000 prices and euros.

The figures are reasonably consistent, although it is unclear why ERV estimates for Portugal and the UK are so high.

10 Conclusions

Those using economic valuations of risks due to accidents and pollution face the options of commissioning more original ('primary') studies or adopting a benefits transfer procedure. In the context of risks to life and health transfer procedures remain problematic, even though they are probably the most widely used methods in current cost-benefit studies in Europe. The problems arise from the general failure to conduct meta-studies of the estimates being derived from primary studies. In quite a number of cases the absence of a meta study simply reflects the limited number of primary studies. In others it reflects the failure of the primary studies to control for many contextual features, including fairly obvious one such as age and even risk level. In these circumstances, a recommendation that substantially more primary studies are needed is unavoidable.

In the context of premature mortality, there is a general failure to investigate the impact of context on willingness to pay. This situation is changing with very recent work which does suggest that it is legitimate to adjust WTP typical of a 40 year old downwards for those over 70 years of age. The adjustment for pollution as opposed to accidents is generally unknown, however, whilst suggested a priori adjustments - e.g. downwards for state of health - are not borne out by the limited evidence. The only available meta studies all relate to wage risk studies and there are theoretical concerns about what exactly these studies are measuring, and, even if they are valid measures of WTA, whether they can be transferred to other contexts. Fairly popular 'conversions' of VOSLs into VOLYs appear to have no theoretical basis. At the moment, there appears to be little basis for a 'disaster aversion' factor for large group accidents but this may not be consistent with some of the psychometric literature which does suggest aversion to certain kinds of accidents. The VOLY issue does, however, raise the matter of just what lifetime reduction is being suffered by pollution victims. Some recent work suggesting extremely small reductions in expected lifetime needs to be developed. Overall, then, the situation with respect to the appropriate VOSL which can be adjusted for context is not a good one.

For morbidity the picture is better, largely due to the multi-country European study of morbidity end points, with some control for context in these sense of causal factors. This suggests that transfers may be made with errors of perhaps + or - 40%. This work requires extension to cover more types of morbidity endpoint, and especially cancer contexts which are relevant to many pollution events.

References

AEA Technology, 1998a. *Cost-Benefit Analysis of Proposals Under the UNECE Multi-Pollutant Multi-Effect Protocol*, Report to UK Department of Environment, Transport and Regions, London and to UNECE Task Force on Economic Aspects of Abatement Strategies, Geneva.

AEA Technology, 1998b. *Economic Evaluation of the Control of Acidification and Ground Level Ozone*, (provisional), Report to DGXI of the European Commission, Brussels.

AEA Technology, 1998c. *Economic Evaluation of Air Quality Targets for CO and Benzene*, DGXI, European Commission, Brussels.

AEA Technology 1999. *Cost-Benefit Analysis for the Protocol to Abate Acidification, Eutrophication and Ground Level Ozone in Europe*, Ministry of Housing, Spatial Planning and the Environment, The Hague, Netherlands, Publication No. 133.

Aimola, A. (1998), Individual WTPs for reductions in cancer death risks, in R. Bishop and D. Romano (eds), *Environmental Resource Valuation: Applications of the Contingent Valuation Method in Italy*, Kluwer, Dordrecht, 196–212.

Ascari, S and Bernasconi, M. 1997. *The Economics of Risk and Uncertainty and the Valuation of Severe Accidents*, Paper prepared for the ExternE programme.

Bergstrom, T. 1982. When is a man's life worth more than his human capital? in M. Jones-Lee (ed), *The Value of Life and Safety*, North Holland, Amsterdam, 3-26.

Bleichrodt, H and Quiggin, J. 1999. Life-cycle preferences over consumption and health: when is cost-effectiveness analysis equal to cost-benefit analysis? *Journal of Health Economics*, **18**, 681-708

Blomquist, G., Miller, T. and Levy, D. 1996. Values of risk reduction implied by motorist use of protection equipment, *Journal of Transport Economics and Policy*, January, 55-66.

Brouwer, R. 1998. *Future Research Priorities for Valid and Reliable Environmental Value Transfer*, Paper GEC 98-28, Centre for Social and Economic Research on the Global Environment, University of East Anglia and University College London, London

Brouwer, R and Spanincks, F. 1999. The validity of environmental benefits transfer: further empirical testing, *Environmental and Resource Economics*, **14**, 95-117.

CEC (Commission of the European Communities) DGXII, 1995. *ExternE: Externalities of Energy*, Vols 1-6, CEC, Brussels

CEC (Commission of the European Communities) DGXII, 1998. *ExternE: Externalities of Energy*, Vols 7-10, CEC, Brussels

Cropper, M., Aydede, S and Portney, P.1994. Preferences for life saving programs: how the public discounts time and age, *Journal of Risk and Uncertainty*, **8**, 243-265

Cropper, M and Sussman, F. 1988. Families and the economics of risk to life, *American Economic Review*, 255-260.

Day, B. 1999. *A Meta-Analysis of Wage-Risk Estimates of the Value of Statistical Life*, Centre for Social and Economic Research on the Global Environment, University College London, London, *mimeo*.

Desaigues, B and Rabl, A. 1995. Reference Values for Human Life: An Econometric Analysis of a Contingent Valuation in France, in N Schwab Christie and N.Soguel (eds), *Contingent Valuation, Transport Safety and the Value of Life*, Kluwer, Boston and Dordrecht, 85-112.

Desvousges, W., Johnson, F.R and Banzaf, H.S, 1998. *Environmental Policy Analysis with Limited Information: Principles and Applications of the Transfer Method*, Cheltenham: Edward Elgar.

Eeckhoudt, L., Schieber, C and Schneider, T. 1997. *Expected Utility Approach: Risk Aversion and Collective Risk*, Paris: CEPN. (Paper prepared for ExternE programme)

Freeman, A.M. 1993. *The Measurement of Environmental and Resource Values: Theory and Methods*, Washington DC: Resources for the Future.

Hammitt, J and Graham, J, 1999. Willingness to pay for health protection: inadequate sensitivity to probability, *Journal of Risk and Uncertainty*, **18**, 33-62

Hansen, A and Selte, H, 2000. Air pollution and sick-leaves, *Environmental and Resource Economics*, **16**, 31-50

Holland, M and Krewitt, W. 1996. *Benefits of an Acidification Strategy for the European Union*, European Commission, DGXI, Brussels.

IEI (Industrial Economics Incorporated), 1992. *Review of Existing Value of Life Estimates*, Document prepared for US EPA, Washington DC.

IVM, NILU and IIASA, 1997. *Economic Evaluation of Air Quality for Sulphur Dioxide, Nitrogen Dioxide, Fine and Suspended Particulate Matter and Lead*, Report to DGXI, European Commission, Brussels.

M.Johannesson and P-O Johansson, 1996. To be or not to be, that is the question: An empirical study of the WTP for an increased life expectancy at an advanced age, *Journal of Risk and Uncertainty*, **13**, 163-174.

M.Johannesson and P-O Johansson, 1995a. *Is the Value of a Life Year Gained Independent of Age ?*, Stockholm School of Economics, Stockholm, *mimeo*.

M.Johannesson and P-O Johansson, 1995b. Quality of life and the WTP for an increased life expectancy at an advanced age, *Working Papers in Economics and Finance, No.85*, Stockholm School of Economics.

Johansson, P-O (1995), *Evaluating Health Risks: an Economic Approach*, Cambridge University Press, Cambridge.

M.Jones-Lee, 1989. *The Economics of Safety and Physical Risk*, Blackwell, Oxford.

Jones-Lee, M., Hammerton, M and Phillips, P. 1985. The value of safety: results of a national sample survey, *Economic Journal*, **95**, 49-72

Jones-Lee, M. 1991. Altruism and the value of other people's safety, *Journal of Risk and Uncertainty*, **4** :2, 213-219

Jones-Lee, M. 1992. Paternalistic altruism and the value of statistical life, *The Economic Journal*, **102**:410, 80-90.

Jones-Lee, M., Loomes, G., O'Reilly, D and Philips, P. 1993. *The Value of Preventing Non-fatal Road Injuries: Findings of a Willingness to Pay National Sample Survey*, Working Paper WP/SRC/2, Transport Research Laboratory, Crowthorn.

Jones-Lee, M and Loomes, G. 1995. Scale and context effects in the valuation of transport safety, *Journal of Risk and Uncertainty*, **11**, 183-203.

Kidholm, K. 1995. Assessing the Value of Traffic Safety Using the Contingent Valuation Technique: the Danish Survey, in N Schwab Christie and N.Soguel (eds), *Contingent Valuation, Transport Safety and the Value of Life*, Kluwer, Boston and Dordrecht, 45-61

Kirchhoff, S., Colby, B and LaFrance, J, 1997. Evaluating the performance of benefit transfer: an inquiry, *Journal of Environmental Economics and Management*, **33**, 75-93

Krewitt, W., Holland, M., Trukenmüller, A, Heck, T and Friedrich, R, 1999. Comparing costs and environmental benefits of strategies to control acidification and ozone in Europe, *Environmental Economics and Policy Studies*, **2**, 249-266.

Krupnick, A., Markandya, A and Nickell, E. 1993. The external costs of nuclear power; ex ante damages and lay risks, *American Journal of Agricultural Economics*, **75**, 1273-1279

Krupnick, A., Harrison, K, Nickell, E and Toman, M. 1996. The value of health benefits from ambient air quality improvements in Central and Eastern Europe, *Environmental and resource Economics*, **7**, 307-332.

Krupnick, A., Alberini, A., Cropper, M., Simon, N., Itaoka, K and Akai, M. 1999. *Mortality Risk Valuation for Environmental Policy*, Discussion Paper 99-47, Resources for the Future, Washington DC.

Krupnick, A., Alberini, A., Cropper, M., Simon, N., O'Brien, B., Goeree, R and Heintzelman, M. 2000. *Age, Health and Willingness to Pay for Mortality Risk Reductions: A Contingent*

Valuation of Ontario Residents, Discussion Paper 00-37, Resources for the Future, Washington DC.

Loehman, E., Berg, S., Arroyo, A., Hediger, R., Schwarz, J., Shaw, M, Fahien, W, De, V., Fische, R., Rio, D., Rossley, W and Green, A. 1979. Distributional analysis of regional benefits and cost of air quality control, *Journal of Environmental Economics and Management*, **6**, 222-43

Lutter, R. 1998. *An Analysis of the Use of EPA's Benefit Estimates in OMB's Draft Report on the Costs and Benefits of Regulation*, Comment 98-2, AEI-Brookings Center for Regulatory Studies, Washington DC.

Maddison, D. 1998. *Valuing Changes in Life Expectancy in England and Wales Caused by Ambient Concentrations of Particulate Matter*, Paper GEC 98-06, Centre for Social and Economic Research on the Global Environment, University College London, London

Maddison, D. 1999. *The Plausibility of the ExternE Estimates of the External Effects of Electricity Production*, Paper GEC 99-04, Centre for Social and Economic Research on the Global Environment, University College London, London.

Maddison, D. 2000a. *Valuing the Morbidity Effects of Air Pollution*, Centre for Social and Economic Research on the Global Environment, University College London, London, *mimeo* (see also UK Department of Health, 1999)

Maddison, D. 2000b. *Modelling Distributed Lag Effects in Epidemiological Time Series Studies*, Centre for Social and Economic Research on the Global Environment, University College London, London, *mimeo*

Maier, G., Gerking, S and P.Weiss, 1989. *The Economics of Traffic Accidents on Austrian Roads: Risk Lovers or Policy Deficit? Mimeo*, Wirtschaftuniversitat, Vienna

McDaniels, T., Kamlet, M and Fischer, G. 1992. Risk perception and the value of safety, *Risk Analysis*, **12**, 495-503

Mendelof, J and Kaplan, R. 1990. Are twenty fold differences in 'lifesaving' costs justified? A psychometric study of the relative value of preventing deaths from programs addressing different hazards, In Cox, L and Ricci, D (eds), *New Risks*, Plenum Press, New York

Miller, T and Guria, J. 1991. *The Value of Statistical Life in New Zealand*, New Zealand Ministry of Transport, Wellington NZ.

Needleman, L.1976, Valuing other people's lives, *The Manchester School*, **44**, 309-342.

Olsthoorn, X., Amann,M., Vartanova, A., Clench-Aas, J., Cofala, J., Dorland, K., Guerreiro, C., Henriksen, J., Jansen, H and Larssen, S. 1999. Cost-benefit analysis of European air quality targets for sulphur dioxide, nitrogen dioxide and fine and suspended particulates in cities, *Environmental and Resource Economics*, **14**, 333-351

Pearce, D.W. 1980. The social incidence of environmental costs and benefits, *Progress in Resource management and Environmental Planning*, **2**, 63-87

Pearce, D.W and Crowards, T. 1996. Particulate matter and human health in the United Kingdom, *Energy Policy*, **24**, 7, July, 609-620

Pearce, D.W., Dubourg, R., Day, B., Atkinson, G., Navrud, S., Ready, R., Kuik, O., Spanincks, F., Labandeira-Villot, X., Vasques Rodrigues, M., Machado, F, and Mourato, S. 1998. *Benefit Transfer and the Economic Valuation of Environmental Damage in the European Union with Special Reference to Health, Final Report: Summary, Final Report, Annexes*, CEC, DGXII, Brussels.

Pearce, D.W and Ulph, D. 1999. A social discount rate for the United Kingdom, in D.W.Pearce, *Economics and Environment: Essays in Ecological Economics and Sustainable Development*, Cheltenham: Edward Elgar, 268-285

Persson, U. and Cedervall, M. 1991. The Value of Risk Reduction: Results of a Swedish Sample Survey, *IHE Working Paper 1991:6*, Swedish Institute of Health Economics.

Persson, U., Ligner Norinder, A and M.Svensson, 1995. Valuing the benefits of Reducing the Risk of Non-Fatal Road Injuries: the Swedish Experience, in N Schwab Christie and N.Soguel (eds), *Contingent Valuation, Transport Safety and the Value of Life*, Kluwer, Boston and Dordrecht, 63-83

Pope, C., Thun, M., Namboodiri, M., Dockery, D., Evans, J., Speizer, F and Health, C. 1995. Particulate air pollution as a predictor of mortality in a prospective study of US adults, *American Journal of Respiratory Critical Care Medicine*, **151**, 669-674.

Reed Johnson, F., Fries, E and Banzaf, H, 1996. *Valuing Morbidity: An Integration of the Willingness-to-Pay and Health-Status Literature*, TER Technical Working Paper &-9601, Triangle Economic Research: Durham, North Carolina

Römer, A., Pommerehne, W and Feld, L. 1998. Revealing preferences for reductions of public risks: an application of the CV approach, *Journal of Environmental Planning and Management*, **41** (4), 477-503

Rowe, R., Lang, C., Chestnut, L., Latimer, D., Rae, D., Bernow, S. *et al.* (1995), *The New York Electricity Externality Study*, New York: Oceana Publications.

Rowlatt, P., Spackman, M., Jones, S., Jones-Lee, M and Loomes, G. 1998. *Valuation of Deaths from Air Pollution*, NERA and CASPAR for UK Department of the Environment, London.

Savage, I. 1993. An empirical investigation into the risks of psychological perceptions on the willingness to pay to reduce risk, *Journal of Risk and Uncertainty*, **6**, 75-90

Schwab Christie, N and Soguel, N. 1995, *The Pain of Victims and the Bereavement of their Relatives: a Contingent Valuation Experiment*, IDHEAP, University of Lausanne, Switzerland, mimeo.

Sieg, H., Smith, V.K., Banzaf, H.S and Walsh, R. 2000. *Estimating the General Equilibrium Benefits of Large Policy Changes: the Clean Air Act Revisited*, National Bureau of Economic Research, Working Paper 7744, NBER, Cambridge, Mass.

Smith, V.K and Desvousges, W. 1987. An empirical analysis of the value of risk changes, *Journal of Political Economy*, **95**, 89-113.

UK Department of Health, 1999. *Economic Appraisal of the Health Effects of Air Pollution*, Ad-Hoc Group on the Economic Appraisal of the Health Effects of Air Pollution (EAHEAP), London.

UK HM Treasury, 1997. *Appraisal and Evaluation in Central Government: the 'Green Book'*, London: HM Treasury

UNDP (United Nations Development Programme), 2000. *The Human Development Report 2000*, Oxford: Oxford University Press, Oxford

US Environmental Protection Agency, 1997. *The Benefits and Costs of the Clean Air Act 1970-1990*, USEPA, Washington DC.

Van den Bergh, J., Button, K., Nijkamp, P and Pepping, G. 1997. *Meta-analysis in Environmental Economics*, Dordrecht: Kluwer

Viscusi W.K. (1995), The automobile risk metric for valuing health risks, in N. Schwab, Christie and N. Soguel (eds), *Contingent Valuation, Transport Safety and the Value of Life*, Kluwer, Dordrecht, 171–93.

Zuidema, T and Nentjes, A, 1997. Health damage of air pollution: an estimate of a dose response relationship for the Netherlands, *Environmental and Resource Economics*, **9**, 291--308