EPOC HIGH-LEVEL SPECIAL SESSION ON THE COSTS OF INACTION:

THE COSTS OF INACTION WITH RESPECT TO HUMAN HEALTH IMPACTS FROM POLLUTION:

BACKGROUND PAPER PREPARED BY STÅLE NAVRUD (CONSULTANT)

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The Costs of Inaction with Respect to Human Health Impacts from Pollution[1]

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1. Introduction

1. In many environmental regulation contexts, an important category of impacts from policy inaction is impacts on public health. These can include impacts on both rates of ill-health (morbidity)[2] and rates of fatal illnesses (mortality) in the affected population. Indeed, for regulations aimed at improving air or water quality, health benefits can be the dominant category of impacts in a regulatory impact analysis.

2. Studies of external costs of energy (European Commission, 1999) and transportation (Friedrich and Bickl, 2001) within the ExternE project series[3] of the European Commission show that human health impacts from air pollution make up a very significant part of the costs of environmental policy inaction. The aggregated damages of electricity generation were in othe order of about 1% of Gross Domestic Product (GDP) in most EU15-countries. In a prospective cost-benefit analysis of the 1990 Clean Air Act Amendments (US EPA, 1999), decreases in mortality and morbidity from improved air quality constituted 96% of the total estimated benefit. A recent Polish study (Dziegielewska and Mendelsohn, 2005) found health costs represented 61-77% of total costs of air pollution. Dziegielewska and Mendelsohn (2005) also found that the costs of air pollution policy inaction compared to a 25-50% reduction in premature deaths from reduced air pollution, was in the order of 0.8-1.0% of Gross Domestic Product (GDP).

3. The main purpose of this paper is to review work and evidence to date of the welfare loss from human health impacts from selected pollutants. The review will focus on the impacts of air pollutants, especially particulate matter (PM), and the impacts on morbidity (i.e. acute and chronic respiratory illnesses) and mortality (i.e. premature deaths). Section 2 will provide an overview of the Damage Function Approach to the valuation of the costs of air pollution. Sections 3 and 4 will review existing valuation studies on morbidity and mortality, respectively. Methodological difficulties associated with the valuation of these impacts will be discussed in both these sections, and summarized in section 5, while section 6 concludes with a summary of the implications for policy makers, including interim recommendations for unit values and how to treat critical methodological issues, and proposals for future work in this area.

2. The Damage Function Approach

4. The damage function approach (DFA)[4] is the general approach used to assess the economic value of health impacts; see Box 1 for an overview of the four steps of the DFA applied to public health impacts of air pollutants.

[1] In accepting that the document be made more widely available, some member countries have significant concerns with the content of this paper.
[2] Ill health outcomes (called endpoints in the environmental epidemiology literature) might be an episode of ill health such as a minor symptom day or a hospital admission, or it might be a case of a disease, such as chronic bronchitis or asthma.
[3] See the ExternE project series website for further details: http://www.externe.info/, and the European Commision DG Environment website on environmental economics for cost-benefit analyses of air quality Targets and Directives for several air pollutants: http://europa.eu.int/comm/environment/enveco/studies2.htm.
[4] Termed the Impact Pathway Approach by the ExternE project series of the European Commission.

5. The approach generally used to value decrements in health resulting from environmental policy inaction is somewhat unique, in that it relies heavily on unit values and value transfer. The typical approach when valuing environmental health decrements from policy inaction is to first predict the increase in health outcomes as a result of the inaction. These predictions are made by combining projected changes in emissions and exposure to pollutants with established exposure-response relationships (or expert asessmemts), see Appendix 3; i.e. steps 1, 2 and 3 in Box 1. This type of analysis gives predictions of the increase in morbidity and mortality outcomes as a result of the inaction. In step 4 these impacts on public health are valued by multiplying the number of each type of morbidity and mortality outcome by a constant value specific to each outcome.

6. This paper focuses on the fourth step in this approach, i.e. multiplication of the number of health outcomes by an outcome-specific unit value per incidence and the uncertainty of these estimates. However, one should keep in mind that uncertainty is aggregated over all four steps of the DFA due to uncertainties in atmospheric dispersion models, exposure-response functions (e.g. linking concentrations of particulate matter and premature deaths and symptom days for respiratory illnesses; see WHO (2000, Box 1) and European Commission (1999) for lists of potentially relevant acute and chronic mortality and morbidity impacts from air pollution, and the health valuation techniques and techniques used to transfer these monetary values spatially and temporally (Navrud, 2004).

Box 1. The Damage Function Approach (DFA) applied to emissions of air pollutants

Step 1: *Emissions of pollutants*

Step 2: → \Atmospheric Dispersion Model\ → *Changed concentrations and exposure*

→ \Exposure-response functions (see Appendix 3)\ and

\Size and characteristics of the affected population\

Step 3: → *Number of mortality and morbidity outcomes*

Step 4: → \Unit values for each outcome from Benefit Transfer techniques\;

summed up over all outcomes; or new valuation study for

each outcome or the overall program or policy inaction.

→ *Social Costs of Policy inaction*

\ --- \ = models

--- = output (or input)

7. Environmental pollution that impairs human health can reduce people's well-being through at least the following five channels:

1. medical expenses associated with treating pollution-induced diseases, including the opportunity cost of time spent in obtaining treatment;
2. lost wages;
3. defensive or avertive expenditures associated with attempts to prevent pollution-induced disease;
4. disutility associated with the symptoms and lost opportunities for leisure activities; and,
5. change in life expectancy or risk of premature death.

8. To obtain an estimate of the social costs of health impacts from air pollution, we need to map all these categories of costs, including costs both to the affected individuals as well as costs to the employers in terms of lost work days and productivity loss, and the medical costs covered by the public health care system and medical insurance companies. The first three of the five categories listed above have readily available monetary counterparts; and the first two constitute Cost-of-Illness (COI) measures. However, a truly comprehensive measure of social costs of policy inaction with regards to health should capture all these categories, as measures based solely on medical costs and lost wages as a measure of COI would omit major categories of the costs. Rowe *et al*. (1995, volume 1, table 10-5) review studies comparing Willingness-to-Pay (WTP) and COI, and find WTP/COI ratios for the affected individual (and their family), and for society as a whole, at 1.6-2.3 and 1.3-1.7, respectively for asthma (which is the endpoint of exposure-response functions for air pollutants, especially particulate matter). Reviewing the evidence from two other studies, they conclude that the social WTP/COI ratio ranges from 1.3 to 2.4, and use 2.0 as a central value for all health impacts from air pollution, for which they have no WTP estimaters, except cancer where they use 1.5 for a nonfatal case (mainly due to the high and more uncertain COI for cancer). Similar ratios have been used for morbidity valuation both in the ExternE (External costs of energy) project series (European Commission, 1999), as well as in the Methodology Report for the Cost Benefit Analysis (CBA) of the Clean Air for Europe (CAFE) – program (Holland *et al*., 2004).

9. *Mortality* is most often valued in terms of Value of a Statistical Life (VSL), which is the rate at which people are prepared to trade off income for a reduction in their risk of dying. There are two basic nonmarket valuation approaches suggested for identifying the willingness to pay of an individual for mortality risks. Firstly, the Hedonic Wage (HW) method, which is a Revealed Preference (RP) method, observing actual behaviour in the labour market. If a person is working in a job with above average mortality risk then they will require a higher wage to compensate for this risk. By observing the wage premium, we can see what value they attach to that risk. One drawback of hedonic wage studies is that they provide estimates of VSL only for a small segment of the population. A second shortcoming is that these studies value current risk of accidental death, whereas environmental hazards, such as asbestos or PCBs, are likely to cause death after a latency period, with cause of death being cancer or chronic respiratory illness.

10. Secondly, Stated Preference (SP) studies explicitly ask individuals how much they would be willing to pay (or willing to accept) to compensate for a small reduction (increase) in risk. The SP methods can be divided into direct and indirect approaches. The direct Contingent Valuation (CV) method is by far the most used method, but over the past few years the indirect approach of Choice Modeling (CM) has gained popularity. The main difference between these two approaches is that the CV method typically asks the respondent for their willingness-to-pay (WTP) to introduce a public program that would reduce their mortality risk directly as an open-ended question or as a two-alternative (referendum) approach. CM on the other hand, presents the respondents with a series of choices between health risks with different characteristic and monetary amounts. The main appeal of SP methods is that, in principle, they can elicit WTP from a broad segment of the population, and can value causes of death that are specific to environmental hazards.

11. There are practical difficulties with obtaining values from both approaches. In addition, how a person values risk could vary with many factors – age, income and the type of risk itself (e.g. whether the risk is voluntary or involuntary) – and this needs to be reflected in the values.

12. SP methods, especially Contingent Valuation, are also the main method for valuing *morbidity* (ill health). Morbidity can be classified in several ways. One classification is according to duration: *chronic versus acute*. Acute morbidity refers to illnesses that last no longer than a few days and have well-defined beginnings and ends. Chronic morbidity refers to illnesses that are longer term and last indefinitely. Another way of classifying morbidity is by the degree of *impairment of activity*. A Restricted Activity Day

(RAD) is a day in which a person is able to undertake some, but not all, of his/her normal activities. Bed disability days are those on which a person is confined to bed, either at home or in an institution, for all or most of the day. Chronic respiratory symptoms or episodes of illnesses due to air pollution, e.g. coughing days, respiratory emergency room visits and hospital admissions. These acute symptoms are typically valued in a context of certainty, but SP methods can also be used to value the risk of chronic illnesses like lung cancer, asthma and chronic bronchitis.

13. Another approach to valuing both mortality and morbidity risk is the self-protection (or aversive behavior) approach. Here, expenditures people make to reduce either the probability of a bad outcome or severity of the bad outcome are usually assumed, under certain plausible conditions, to be a lower bound on the ex ante value people assign to reduced risks to life and limb. However, recent analysis (Shogren and Stamland, 2005) find that VSL estimated from this method is not in general a lower bound on the population average WTP for mortality risk reduction. Situations arise in which these expenditures are upper bounds, and situations exist when this "lower bound" is a severely deflated lower bound. The economic circumstances describing these situations, unfortunately only partly depend upon things we can observe and correct for, e.g. the fraction who purchase self-protection and the price-setting in the market for self-protection. The impacts of these observable factors are tangled with the impacts of elements we cannot directly observe, e.g. the heterogeneity of both skill to cope with risk and risk preference among people. Thus, more research is still needed to define and broaden the case where one can at least say whether self-protection expenditures are a lower bound of true value, or one is confident of the direction of the bias of a biased (i.e. relatively invalid) value (Bishop, 2003).

14. In practical policy analysis, limited time and resources prohibit new health valuation studies to be performed for each policy decision, and decision makers have to rely on transfer of economic estimates from previous studies of similar changes in acute and chronic mortality and morbidity. There are two main approaches to value transfer (which is a more general term than the often used term "benefit transfer"): i) unit value transfer (i.e. simple unit transfer and unit transfer with income adjustments), and ii) function transfer (i.e. value function and meta analysis). Simple unit transfer is the easiest approach to transferring value estimates from one site to another. This approach assumes that the well-being experienced by an average individual at the study site (where the original valuation study was conducted), is the same as that which will be experienced by the average individual at the policy site. Thus, we can directly transfer the mean value estimate (e.g. mean willingness to pay (WTP) per person to avoid one respiratory symptom day or episode) from the study site to the policy site. The simple unit transfer approach is not fit for transfer between countries with different income levels and standards of living. Therefore, unit transfer using Purchasing Power Parities (PPPs)[5] has been applied. However, this adjustment will not account for differences in preferences, baseline environmental and health conditions, and cultural and institutional conditions which differ between countries. To do this, we need to transfer the entire value function, and plug in values for the explanatory variables at the policy site. This approach is conceptually more appealing than just transferring unit values because more information is effectively transferred. The main problem with this approach arises from the exclusion of relevant variables in the value function estimated in a single study. However, results from several valuation studies can be combined in a meta-analysis to estimate one common benefit function. Meta-analysis has been used to synthesize research findings and improve the quality of literature reviews of valuation studies to come up with adjusted unit values.[6]

[5] For GDP PPPs for all OECD member countries: see http://www.oecd.org/dataoecd/61/54/18598754.pdf

[6] In a meta-analysis original studies are analysed as a group, where the results from each study are treated as a single observation into new analysis of the combined data set.

3. Review of Morbidity Valuation Studies

15. Three categories of value are generally considered for valuing morbidity impacts: 1) the social costs of providing medical treatment to the victim of the ill health outcome; 2) lost labor productivity resulting from the ill health outcome; and 3) the pain, discomfort, and inconvenience suffered by the victim. Per-incidence estimates of the first category of these costs are assembled from hospital records, records of visits to doctors' offices, records of prescription medication use, and surveys of victims of their out-of-pocket health care costs. Per-incidence estimates of lost productivity are usually based on the hourly wages paid to the victim, relying on the theoretical assertion that wages should reflect the marginal value of the victim's labour to his or her employer.

16. Estimation of the third category of value, the pain, discomfort, and inconvenience suffered by the victim, is more problematic, because there are few market prices or financial records that will reveal this value. Instead, the usual approach is to use stated preference techniques such as contingent valuation or stated choice approaches to estimate the victim's willingness to pay (WTP) to avoid an ill health outcome.[7]

17. What makes ill health valuation unique among situations where nonmarket valuation techniques are applied is the implicit assumption that all cases of an ill health episode have the same value. In particular, it is usually assumed that the value of an ill health outcome does not depend on: 1) the cause of that ill health outcome (so that, for example, a day suffering from itchy eyes and a stuffy nose caused by air pollution is valued the same as a similar episode caused by contaminated water at a swimming beach), 2) whether individuals in the population will avoid at most one incidence of an ill health outcome, or whether some individuals will avoid more than one (so that, for example, the value to an individual of avoiding 7 incidences of ill health is 7 times the value of avoiding one incidence), 3) the health status of the individuals who will enjoy improved health (so that the value of avoiding an incidence of ill health to a person with chronic health problems is the same as the value to a person who rarely experiences ill health), and 4) whether ill-health affects children or adults.

18. We will now review the evidence on the validity of using constant per-episode and per-case values when valuing changes in public health due to changes in environmental quality. A second issue that will be explored is the validity of transferring health values estimated in one geographic region to an analysis conducted in another region. This is of pratical importance since relatively few environmental morbidity valuation studies have been conducted, especially outside the United States. Ill-health values are routinely transferred between countries, with little guidance on how values might differ due to differences in health status, socio-economic conditions, or culture.

Valuing One Episode Versus Many Episodes

19. At least for less-serious ill health outcomes, it is common practice in stated preference studies valuing health to value a discrete, marginal change in the number of episodes or cases of ill health that the respondent will experience, rather than valuing a change in risk of ill health. This approach is clearly unrealistic – future health cannot be guaranteed. Further, a risk-free treatment that focuses on health outcomes, rather than on risks, does not allow for the consideration of potential changes in defensive actions that the respondent might take, such as limiting activity during periods of poor air quality. On the other hand, valuing changes in risk imposes difficulties on both the respondent and the researcher. For this reason, most morbidity valuation studies have measured WTP to avoid, with certainty, one or more specific episodes or cases of ill health.

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[7] For a review of US empirical estimates of ill health caused by pollution, see US EPA 1999, Appendix H.

20. While exposure-response studies may tell us how many fewer hospital admissions and minor symptom days will occur as a result of an improvement in environmental quality, they usually do not predict how these avoided outcomes will be distributed within the affected population. For many environmental health issues, there is an at-risk sub-population that suffers a disproportionate share of the total number of ill health outcomes. For example, asthma attacks are concentrated among those who have asthma. The health improvement that results from an improvement in environmental quality will likewise be concentrated within the susceptible sub-population, and individual sufferers who benefit may avoid more than one episode or case as a result of the policy action. Does the value of avoiding a single episode of ill health depend on how many episodes the individual will avoid? The evidence is that it does.

21. However, to date, the empirical evidence on whether marginal WTP to avoid ill health increases or decreases as the duration of the ill health increases is mixed, though results consistent with declining marginal disutility of ill health (Tolley *et al*., 1994; Johnson *et al*., 2000; Navrud, 2001) are more common than results consistent with declining marginal utility of good health. Complicating these results is the possibility that the elicitation methods used may be unable to reliably measure how value changes as the scope of the health improvement changes. At a minimum, the evidence to date suggests that it is inappropriate to assume that marginal WTP per outcome avoided is constant regardless of the number of outcomes avoided by each individual.

Do People with Poor Health Value Health Differently from those Without Health Problems?

22. Related to the issue of how many outcomes an individual avoids is the issue of who in the population avoids the ill health outcomes. If it tends to be persons with poorer health who lose most from environmental policy inaction, then it is of interest to know whether marginal WTP to avoid one ill health outcome varies with the individual's health status.

23. Tolley *et al*. (1994) report conflicting results as to whether health status affects WTP to avoid days of ill health. WTP to avoid one day of minor symptoms was generally positively related to the number of days the respondent experienced those symptoms within the past 12 months, and was negatively related to overall indicators of health. However, WTP to avoid 30 days of minor symptoms or to avoid 10 or 20 days of angina were not related to health status. Dickie *et al*. (1987) found that WTP to avoid one day of nine different symptoms that can be caused by ozone exposure was not sensitive to how often respondents experienced the symptoms, or whether respondents were respiratory impaired. Johnson *et al*. (2000) found that WTP to avoid episodes of respiratory and cardiac ill health was higher for respondents who had been diagnosed with cardiovascular or respiratory conditions, or other serious illness.

24. There are several studies where WTP to avoid ill health outcomes was higher for respondents who suffered from that type of outcome more frequently, or respondents with poorer health measured more generally, while there are very few results that showed the opposite result. Ready *et al*. (2004a) found that WTP to avoid four different episodes of respiratory illnesses and running eyes (that are endpoints in exposure response functions for air pollutants like particulate matter). We conclude that a weak negative relationship probably exists between health status and WTP to avoid ill health for most ill health outcomes caused by air pollution.

Does The Cause of the Ill Health Matter?

25. Many ill health outcomes that are caused by one type of pollution could be caused by other types of pollution as well (or by other non-environmental factors as well). Nausea, for example, can be caused by air pollution, contaminated drinking water, contaminated swimming beaches, food-borne disease, or by person-to-person transmission of disease. Does the value of avoiding an ill health outcome depend on the cause of that outcome?

26. Few studies have examined this issue directly. Most environmental health valuation studies are deliberately vague about the cause of the prospective ill health, or the mechanism by which their health would be improved. The fear is that if respondents were told that the health improvement would be delivered by an improvement in environmental quality, they would include in their WTP values the co-benefits (improvements in visibility, ecological services, etc.) that would logically result from the environmental quality improvement, making determination of a value-per-day or a value-per-episode difficult. Indeed, where WTP values are measured without reference to the cause of the ill health are compared to WTP values for the same health improvement brought about by an improvement in environmental quality, the latter are found to be larger than the former (Rozan and Willinger, 1998).

27. Ready *et al*. (2004b) attempted to isolate the impact of the cause of ill health on its value, without confounding the value with consideration of how the improved health would be delivered. In five European countries, WTP to avoid six specific episodes of ill health linked to air and water pollution was measured. All of these episodes could be caused either by poor air quality or by swimming at contaminated beaches. A split sample design was used, where some respondents were told the cause of the prospective ill health (air pollution or contaminated water) and others were not told the cause. Neither group was told how the ill health would be avoided. Rather, as is common in this literature, respondents were told that by paying a specified sum, they could avoid one episode with certainty. The results showed no significant difference in WTP between the two samples. This result gives some comfort that the common practice of applying per-incidence values, without consideration of the specific cause of the ill health outcome, is valid.

Same Morbidity Value in All Countries?

28. Most environmental health valuation studies done to date have been conducted in the United States, though several European studies have been completed more recently. Is it valid to take WTP values for avoided ill health outcomes estimated in one country and use them to value health improvements in a different country (i.e. unit value transfer)? What types of adjustments should one make when making inter-country value transfers?

29. The issue that has received the most attention when making inter-country transfers is differences in wealth. If health is a normal good, then WTP for improvements in health should increase with wealth. Indeed, most empirical studies find that within samples of respondents, WTP is positively related to the respondent's income. When using health values estimated in one country (the study country) in a policy analysis in a second country (the target country), it is logical, then, to suppose that WTP should be adjusted to reflect differences in mean income between the two countries. This is of particular importance when transferring unit values estimated in a developed country to a policy analysis in a less-developed country.

30. There are two common approaches to adjusting WTP values to account for differences in income. First, unit values (WTP values for specific health outcomes) from the study country can be adjusted by assuming a constant income elasticity of WTP. While an assumed income elasticity of 1 may be intuitive, empirical evidence from single-country studies is that the income elasticity of WTP tends to be positive, but less than one. The second approach is to use value functions estimated in the study country to predict WTP in the target country. This approach, called value function transfer, accounts for differences in not only income, but any other characteristic that was measured for each respondent in the original study, and is measurable in the target country. The value function transfer approach relies on the assumption that the two countries share a common value function.

31. To test whether any of the three transfer methods (unit value transfer, unit value transfer with adjustment for income differences, or value function transfer) is valid, it is necessary to measure WTP for the same health improvement in two different countries. Alberini *et al*. (1997) measured WTP to avoid an

episode of acute respiratory illness (caused by air pollution) in Taiwan, and compared values for standardized outcomes to values previously estimated in the U.S. They transferred U.S.-estimated unit values adjusting for income differences between the U.S. and Taiwan, assuming income elasticity of 1 or of 0.4, and compared the transferred values to WTP values estimated in Taiwan. They also transferred a value function estimated in Taiwan to predict WTP in the U.S., and compared those predictions to values previously estimated in the U.S. They could not conclusively state whether one of the three approaches outperformed the others, in part because variation in estimated U.S. WTP values was about as large as variation between Taiwan and the U.S. A further complication is that the U.S. studies did not value exactly the same episodes of ill health as were valued in Taiwan. Alberini and Krupnick (2002) conclude from their value transfer comparison, that assuming an income elasticity of WTP of 1.0, or even making other adjustments, do not appear to be reliable for valuing morbidity and mortality risks in developing countries.

32. Similarly, Chestnut *et al*. (1997) compared WTP to avoid one respiratory illness day estimated in Bangkok, Thailand, with estimates from previous studies conducted in the U.S. They found that, even though average income in Bangkok is about one-quarter that in the U.S., mean WTP was roughly equal in the two countries. Again, interpretation of this result is complicated by the fact that the U.S. and Bangkok studies used different survey instruments.

33. When we lack data on the income levels of the affected populations at the policy and study sites, Gross Domestic Product (GDP) per capita figures have been used as proxies for income in international benefit transfers. However, Barton and Mourato (2003) clearly shows how this approach could give wrong results in international value transfers (between Portugal and Costa Rica for the episode STOMACH, table 1, caused by water pollution) when income levels at the study and/or policy site deviates from the average income level in the countries.

34. Using the official exchange rates to convert transferred estimates in U.S. dollars to the national currencies does not reflect the true purchasing power of currencies, since the official exchange rates reflect political and macroeconomic risk factors. If a currency is weak on the international market (partly because it is not fully convertible), people tend to buy domestically produced goods and services that are readily available locally. This enhances the purchasing powers of such currencies on local markets. To reflect the true underlying purchasing power of international currencies, the U.S. International Comparison Program (ICP) has developed measures of real GDP on an internationally comparable scale. The transformation factors are called Purchasing Power Parities (PPPs).

35. The Asian Development Bank manual on economic valuation of environmental impacts (ADB, 1996) provides monetary values for health and environmental impacts that are adjusted in proportion to per capita Gross Domestic Product (GDP). They note that it would be more appropriate to use PPP estimates of per capita GDP because, as noted above, these estimates have been adjusted to reflect a comparable amount of goods and services that could be purchased with the per capita national income in each country. However, even if PPP adjusted GDP figures and exchange rates can be used to adjust for differences in income and cost of living in different countries, it will still not be able to correct for differences in individual preferences, initial environmental quality, and cultural and institutional conditions between countries (or even within different parts of a country).

36. Ideally, a validity test of value transfer between countries should use the same survey instrument in both countries. In this way, variation caused by differences in the survey instrument is eliminated. Ready *et al*. (2001b) estimated WTP to avoid episodes of ill health using the same contingent valuation survey instrument in five different European countries, the Netherlands, Norway, England, Portugal and Spain. The six different episodes valued included two different mild symptom days, a minor restricted

activity day, a work-loss day, a bed day, an emergency room visit, and a hospital admission. Table 1 presents brief synopses of the six episode descriptions.[8]

37. The survey instrument was similar in form to that used by Tolley *et al*. (1994). Respondents were first asked questions about their health status, then asked to rank the episodes in order of severity, then asked their WTP to avoid each episode. Split samples in which the episodes were valued in different order showed no evidence of ordering effects (Ready *et al*., 2004a).

Table 1. Ill-Health Episode Descriptions used in Ready *et al*., (2004a)[9]

Episode Name	Epidemiological End Point	Description
EYES	*1 Mild Symptom Day*	*One Day with mildly red, watering, itchy eyes. A Runny nose with sneezing spells. Patient is not restricted in their normal activities.*
COUGH	*1 Minor Restricted Activity Day*	*One day with persistent phlegmy cough, some tightness in the chest, and some breathing difficulties. Patient cannot engage in strenuous activity, but can work and do ordinary daily activities*
STOMACH	*1 Work-Loss Day*	*One Day of persistent nausea and headache, with occasional vomiting. Some stomach pain and cramp. Diarrhoea at least twice during the day. Patient is unable to go to work or leave the home, but domestic chores are possible.*
BED	*3 Bed Days*	*Three days with flu-like symptoms including persistent phlegmy cough with occasional coughing fits, fever, headache and tiredness. Symptoms are serious enough that patient must stay home in bed for the three days*
CASUALTY	*Emergency Room Visit for COPD and Asthma*	*A visit to a hospital casualty department, for oxygen and medicines to assist breathing problems caused by respiratory distress. Symptoms include a persistent phlegmy cough with occasional coughing fits, gasping breathing even when at rest, fever, headache and tiredness. Patient spends 4 hours in casualty followed by 5 days at home in bed*
HOSPITAL	*Hospital Admission for, COPD, pneumonia, respiratory disease and asthma*	*Admission to a hospital for treatment of respiratory distress. Symptoms include persistent phlegmy cough, with occasional coughing fits, gasping breath, fever, headache and tiredness. Patient stays in the hospital receiving treatment for three days, followed by 5 days home in bed*

Note: COPD = Chronic Obstructive Pulmonary Disease

38. As noted, one issue when comparing WTP values from several countries is the appropriate exchange rate to apply. Ready *et al*. (2004a) argue that local currencies should be converted to a common currency using a PPP adjusted exchange rate. In the context of the contingent valuation survey, improved health is a market good – it is something that gives positive utility that the respondent can choose to buy at a price. The choice whether to purchase the good depends on the respondent's income, the price of improved health, and on the price of other market goods available to the respondent. If two people have identical underlying preferences, but one faces prices that are uniformly α percent higher than those faced by the other, then their behaviour will be identical only if their incomes and the price of improved health

[8] STOMACH (resulting from water pollution) was valued only in England and Portugal. In Spain, the CASUALTY episode lasted only 3 days, and the HOSPITAL episode lasted only 6 days.

[9] All episodes are endpoints in exposure response functions for air quality, with the exception of "Stomach", which is linked to water pollution.

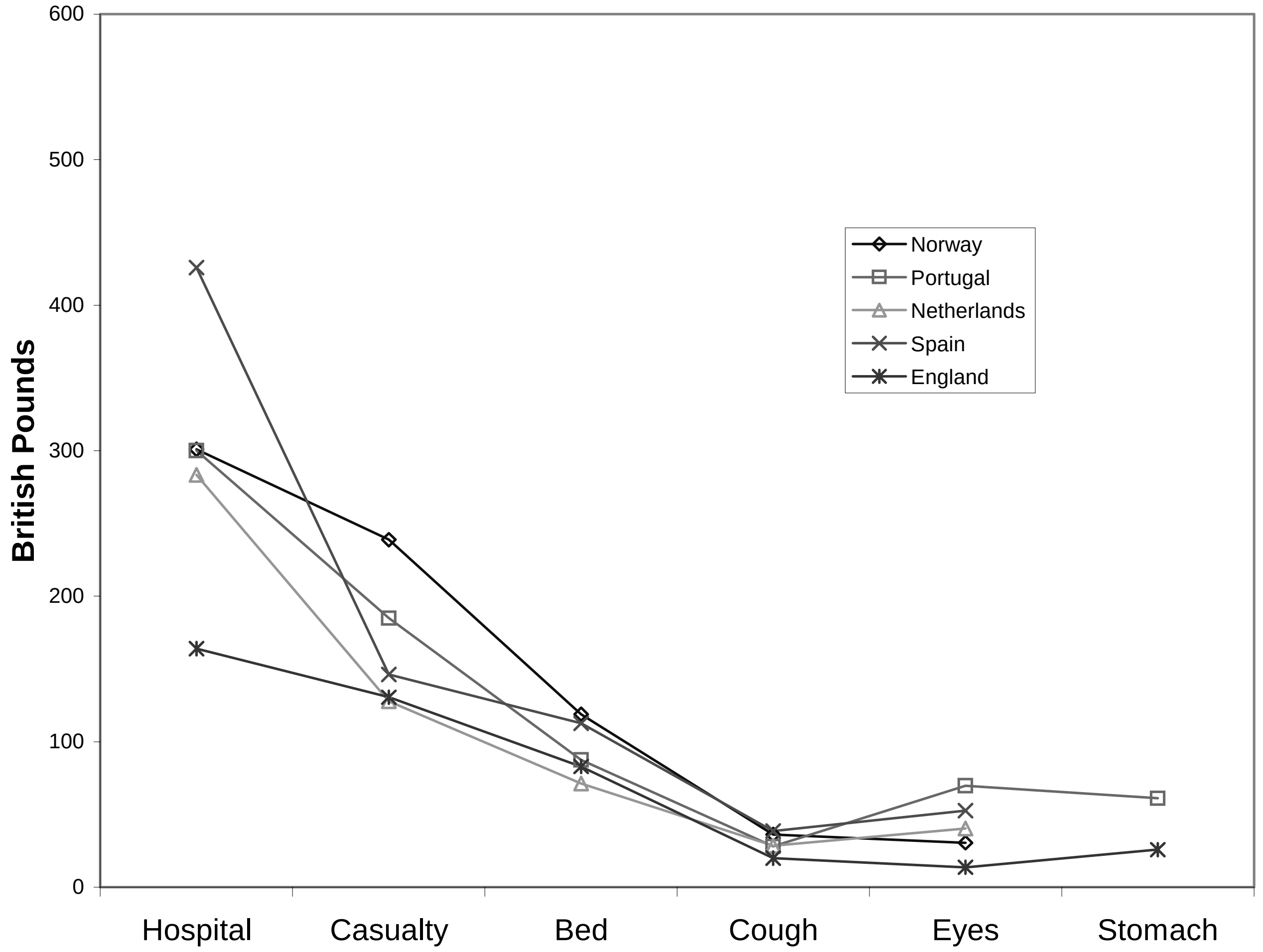
also differ by the same proportion. Following this reasoning, all income and WTP values should be converted into a common currency, using PPP-adjusted exchange rates.

39. Mean WTP values for each episode for each country, converted to GBP, are shown in Figure 1. As would be expected, WTP is higher for the episodes that are more serious and last longer. The three episodes that only last one day, COUGH, EYES, and STOMACH, have the lowest mean WTP values in every country. Comparing results across countries, Norway and Spain have consistently high WTP compared to the other three countries, while England or the Netherlands have consistently low WTP. Portugal tends to have intermediate WTP values, except for EYES, where it has the highest. These apparent differences are in many cases statistically significant.

40. These results are somewhat counterintuitive, given differences in income among the countries. Spain and Portugal have much lower mean real incomes than the three Northern European countries, yet these two countries generally have intermediate to high WTP values relative to the other countries. However, several other differences exist among the countries that have relevance for health valuation (education, family size, current health status). To control for these differences, value functions were estimated for each country, for each episode. Explanatory variables used in the regressions included respondent's income, education level, sex, age, whether there are children in the household, and measures of the respondent's health status and recent experience with symptoms included in the episode descriptions.

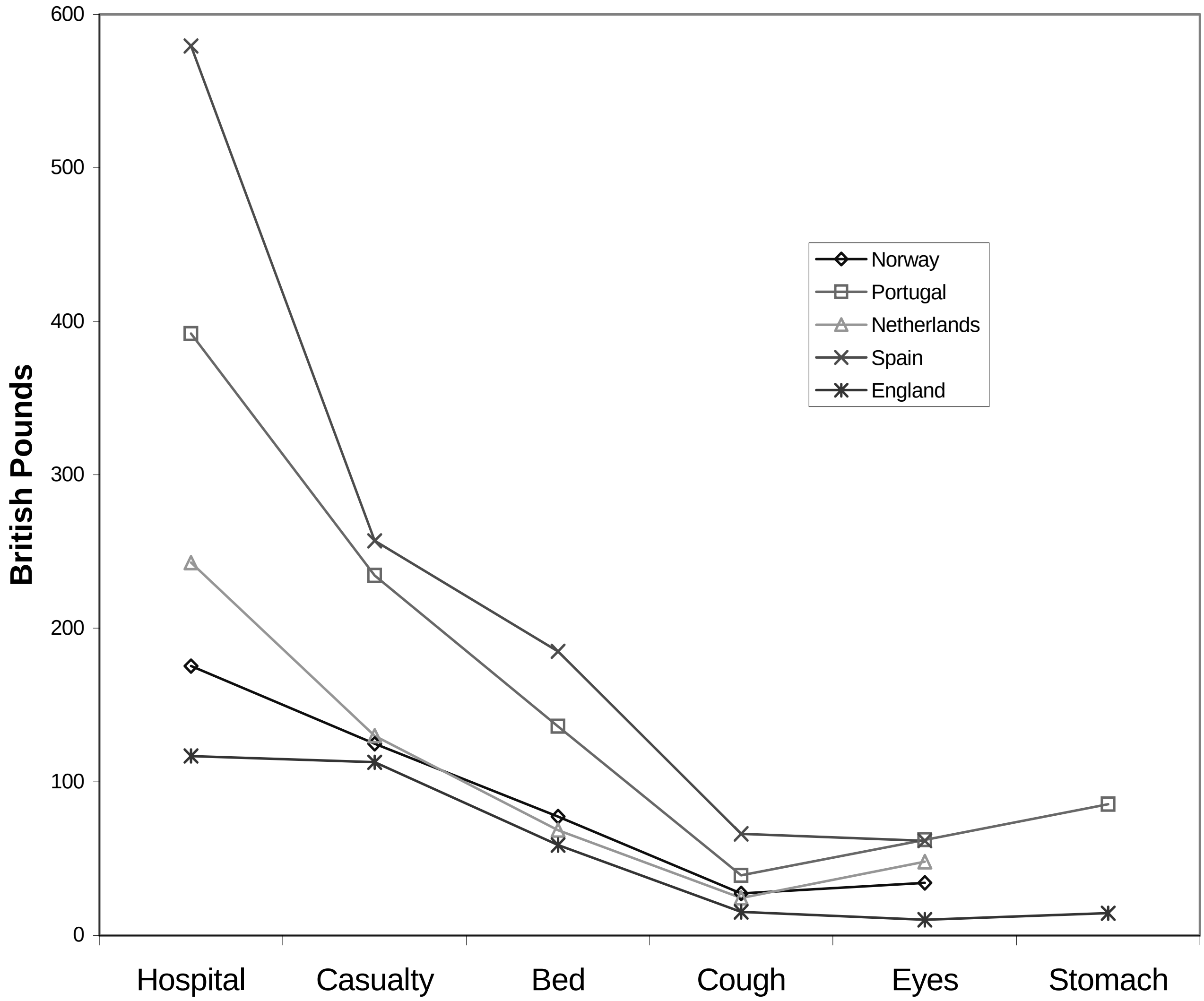
41. Using these value functions, it is possible to construct a WTP estimate for each country for a "standardized" respondent – one who is identical in all measurable characteristics. Figure 2 shows this predicted WTP for each episode for each country, for a respondent with characteristics equal to the mean level for each of the five countries. Here, the pattern of results is more clear. WTP for the standardized respondent is consistently higher in Spain and Portugal than in the Northern European countries. WTP for episodes in Portugal and Spain is significantly higher than WTP in the Netherlands, Norway and England. Differences within each of the two groups were small.

Figure 1 WTP to avoid illness episodes (value per episode)[10]



[10] Ready et al. (2004a)

Figure 2 WTP for a 'Standard' Respondent[11]



42. Even though these results show that unit value transfer and value function transfer are not *statistically* valid between pairs of countries, it is still of interest to know the size of potential transfer errors that might result if transfers were conducted. Value function transfer, with the supposed advantage that it accounts for all measurable differences between the source countries and the target country, actually performed worse than the two unit value transfer approaches.

43. The average transfer errors of 38% include not only the error due to transfer between countries, but also error due to sampling variation both in the study countries and the target country. To give some perspective, a Monte Carlo simulation showed that if the same study was done twice in the same country, the two resulting values would differ by, on average, 16%. Thus, the 38% expected transfer error should be assessed relative to this background level of random sampling error.

[11] Ready et al. (2004a)

44. Two consistent results emerge from the three studies examined here. First, despite expectations based on economic theory, adjustment of values for differences in measurable characteristics does not necessarily improve value transfer. Second, while value transfer and value function transfer may be statistically invalid, they may generate transferred estimates that are reliable enough for policy analyses. Indeed, the errors associated with value transfer may not be much larger than the sampling errors that would result if a new study was conducted in the target country, or than differences in values that result from using different survey instruments in the same country.

Value of Ill-health in Children Versus Adults

45. Navrud (2001, table VII) found respondents in Norway to have 2.0 – 2.5 times higher WTP to avoid an increase in annual number of coughing days (which is an endpoint in air pollution exposure-response functions) among their own children compared to the WTP to avoid adults' coughing days. Also, the marginal value of a coughing day seems to decrease at a lower rate for children than for adults. These results correspond well with results from a US CV study (Dickie and Messman, 2004), indicating that parents value children's illness attributes twice as highly as their own, and appears to reflect parental altruism rather than parent-child differences in initial health or illness costs. Parents WTP to avoid own or child illness increases with income, declines with fertility, increases at a decreasing rate with duration and number of symptoms, and depends on perceived discomfort and activity restrictions. They conclude that current value of assessing morbidity benefits or costs of environmental regulation and inaction may understate substantially the value of children's health, particularly in African-American families. Dickie and Messman (2004; table 8) also provide a good, although incomplete, review of previous acute morbidity valuation research.

4. Review of Mortality Valuation Studies

46. Viscusi and Aldy (2003) document over 60 Hedonic Wage (HW) studies conducted in 10 countries, noting that in the US most of these labor market studies produce estimates of VSL in the range of USD 4-9 million. Appendix 1 provides an overview of mortality valuation studies, applying both HW and CV methods. The recommended VSL value from US EPA (see Appendix 2) is based mainly on HW studies. Transferring these HW studies to the environmental policy context implicitly assumes that the preferences of individuals for income and risk do not vary with context. CV and other SP studies have the potential to circumvent many of the shortcomings that the other approaches are sometimes critized for. CV studies can be used to value mortality risk reductions in many contexts, and are not limited to workplace risks.

47. Chilton *et al*. (2004) estimated WTP for both mortality (as a one year gain in life expectancy in normal and poor health) and morbidity (as avoiding a respiratory hospital admission and a day of breathing) associated with reductions in air pollutants. Rather than assuming that people know the exact magnitude of the risks they face (assumed in the HW studies), well-designed CV studies will educate the respondents about them, and the extent of the risk is spelled out explicitly for them. CV also allows us to survey people negatively affected by environmental policy inaction. This is regarded as a particularly advantageous feature of the method, because these people (e.g. the elderly), are likely to be very different than the population covered in HW studies, and may have different preferences for income and risk.

48. Despite these advantages and the flexibility of CV and other SP techniques, much debate surrounds the estimates of VSL from these surveys. Respondents are not used to dealing with probabilities, especially when risks are very small, and the cognitive burden imposed upon them in the survey – or the failure to communicate risks to them in a meaningful way – may result in undesirable effects. These include failure to distinguish between risk reductions of different sizes (i.e. fails a "scope test") (Hammitt

and Graham, 1999), confusion between absolute and relative risk burden (Baron, 1997), protest responses etc. (Carson, 2000).

49. From the exposure-response functions for mortality we obtain two types of estimate: i) For the acute studies, and the less advanced analysis of chronic effects, an estimate of deaths; and ii) For the more advanced analysis of chronic effects based on the use of life tables, an estimate of life years lost.

50. The recent literature defines two options for valuation of mortality impacts, use of the Value of Statistical Life (VSL), and use of the value of a life year (VLY). An appropriate VSL estimate could be applied directly to the output (number of deaths) from the acute studies. However, to apply the VSL to the result of the chronic function using the methodology that is considered most appropriate (i.e. quantification of life years lost) requires conversion of life years lost to a number of deaths. A similar problem arises with the use of the VLY approach, but here it is the estimated number of deaths from acute exposures that need to be converted, this time to the number of life years lost.

51. Krupnick (2004) in his edited peer review of the methodology proposed for the Cost Benefit Analysis of the Clean Air For Europe (CAFE) programme (Holland, 2004) notes that unlike the VSLs, which are computed from estimates of the WTP for risk reductions using HW or SP surveys, the VLYs have been computed from a VSL estimate, usually from a HW study. The simplest of these approaches is to divide VSL by life-years remaining. From an assumed discount rate, an assumed default life expectancy and an assumption that the value of a life year is constant over the remaining life years, the VLY can be calculated. For the US EPA VSL of USD 6.1 million, the resulting VLY for a 40 year loss in life expectancy (assuming death at 45 in the average HW study) at 3%, would be around USD 270,000.

52. A variety of figures have been suggested in the past to represent the average number of life years lost per individual whose death is associated with chronic exposures to particles (as opposed to the average amongst all those exposed, which is, of course, lower given that not all those exposed will die early as a result of their exposure). One option for converting life years lost to deaths is to investigate the causes of death linked to chronic exposure to particles in the cohort studies. This is dominated by cardio-vascular causes with some lung cancer. The effect on non-malignant respiratory disease is not clear. It is possible to estimate the life years lost associated with death from these causes, and to use such estimates to calculate back to premature deaths.

53. While not common practice in economic assessments of environmental heath impacts, in the health sector, life years lost are often coupled with an assessment of the quality of the life year, following the QALY (Quality Adjusted Life Year) approach (Hammitt, 2002), e.g. to conduct cost-effectiveness analyses of alternative measures to improve the heath status of the general population, or of different treatments for the same disease.

54. Very few studies have asked for WTP estimates for VLY directly. The first direct effort to examine this issue was Johannesson and Johansson (1996), who found a very low VLY. The recent DEFRA study (Chilton *et al*., 2004) performed a CV survey of gains in life expectancy of 1, 3 and 6 months in order to come up with an estimate of a VLY (and at poor and good health). Their study does not pass the scope test, but they argue for using the one month subsample to construct a “best” estimate for VLY of GBP 27,630. Krupnick (2004) also argues that since this study specifically evoked air pollution as the cause, this may have reduced WTP since people may have questioned whether it should be their responsibility to pay for air pollution reductions.

55. Krupnick (2004) notes that the VLY measure does not have the lineage enjoyed by VSL, but it has risen in prominence because it is undeniable that most deaths due to environmental policy inaction would be to elderly, and to treat elderly and non-elderly as equivalent for valuation purposes seemed

inappropriate because so many fewer life-years are lost when elderly die. At the same time, the epidemiological literature is not as robust in life years lost, and the VLY literature is very thin, involving only a few studies that directly ask for WTP for additional life expectancy, e.g. Johannesson and Johansson, 1996; Hamitt and Liu, 2004; and Chilton *et al*., 2004. Therefore, Krupnick (2004) is critical to the suggestion to use VLY in the main analysis, with VSL for a sensitivity analysis, in the CBA of CAFE (Holland *et al*., 2004).

56. Table 3 reports VSL estimates for six countries, based on the same CV survey instrument, showing values in the range of EUR 0.5 – 1.5 million. These values are also close to the interim central value of EUR 1.4 million that DG Environment of the European Commission uses; see Appendix 2 (which stems from an expert workshop organised by the European Commission DG Environment in 2000; see http://europa.eu.int/comm/environment/enveco/others/ recommended_interim_values.pdf).

57. However, Krupnick (2004, p. 32) note that the European applications of the Krupnick *et al*. (2002) survey used the 5 in 1,000 risk change in 10 years (which is equivalent to a 5 in 10,000 annual risk change), but did not ask the 1 in 1000 WTP question first, as was done in the US and Canada. Based on the results in the two latter countries, he predicts that the implied VSLs for this smaller risk change would be 2-3 times larger than for the 5 in 1,000 risk change. The VLYs would be raised by a comparable amounts. Krupnick also protests the use of the median, even if it is a more "robust" statics, as the mean is the appropriate measure to use in CBA, as it aggregates the heterogenity of values in the sample.

Table 3. Value of a Statistical Life (VSL) estimates using the same Contingent Valuation survey instrument in many countries[12]

Country	Median WTP (2002-€)
Canada	506,000
USA	700,000
UK	772,000
Italy	1,448,000
France	958,520
Brazil	1,020,000 – 1,770,000

Notes:

1) Not adjusted for purchasing power parity (PPP)

2) Median values are reported here. The median value of the Weibull distribution is considered to be a more robust estimator. Mean WTP is 2-3 times higher, and should be used as upper end of the range estimate to show the uncertainty.

3) The relatively high Italian value may have been the result the Italian sample not being representative of the Italian population

4) The Brazilian study is based on a sample of middle and upper social class individual residents in Sao Paulo, roughly 69% of the total population (Oritz *et al*., 2004)

[12] VSL estimates using the same Contingent Valuation survey instrument (Krupnick et al, 2002, Alberini et al, 2004) in many countries. Respondents are shown diagrams depicting a reduction in mortality risk of 5 in 1,000. The UK, Italy and France studies make up the study from the EU-project NewExt.

Age Dependency

58. The first study to address the issue of age dependency of VSL's was by Jones-Lee (1989), which examined individuals' WTP for reducing the risk of serious motor vehicle accidents. Based on a central VSL of EUR 4 million at age 40, the age VSL variance was found to have an inverted U-shape. Other supporting evidence for a pattern of VSL declining with age is found in Desaigues and Rabl (1995) and Krupnick *et al*. (2000). Another study is Johannesson and Johansson (1996) who use the contingent valuation method to look at the WTP of different respondents aged 18-69 for a device that will increase life expectancy by one year at age 75.

59. The Johannesson and Johansson (1996) results show an increasing WTP with age – though criticism has been levelled at this study on the basis of its elicitation method and small sample size. This pattern relating to age has also been found in a CVM study by Persson and Cedervall (1991). Pearce (1998) concludes on the basis of a review of the literature that the evidence, such that it is, seems to favour a case for a slow decline of VSL with age.

60. The related issue of the futurity of impact (from latent and chronic mortality air pollution effects) has, as far as I am aware, only been empirically estimated in the Alberini *et al*. studies in North America, (Alberini *et al*., 2001) and the NewExt study (Alberini *et al*., 2004). These studies show that future risk changes are valued lower than immediate risk changes in both the US and Canada, resulting in internal discount rates of 4.6% and 8% respectively. Corresponding numbers for France, Italy and the UK were 5, 6 and 10%, respectively.

Health Status

61. Regarding a relationship between *health status* and VSL, the CV evidence is very limited and inconclusive. The principal studies that have explored this linkage are Johannesson and Johansson (1996) who found that WTP values declined with poorer health status, whilst Krupnick *et al*. (2000) found no significant evidence of a relationship. Since most people affected by air pollution probably are not in normal health, the literature to making VSL and VLY dependent on assessing the validity of adjusting VSL and VLY depending upon health status seems too thin to make credible adjustments in practice.

Context and Degree of Voluntariness

62. The relationship between WTP and *context* is similarly underdeveloped in terms of primary CV studies. The main studies, by Jones-Lee and Loomes (1993, 1995) and Covey *et al*. (1995), reported in Rowlatt *et al*. (1998), consider the road transport accident VSL in relation to those for underground rail accident risks, food risks, risks to third parties living in the vicinity of major airports and domestic fire risks. The perceived involuntariness of the underground rail risk attracted a 50% premium on the road VSL, whilst a 25% discount is attached to the risk of a domestic fire. The latter result was thought to reflect the high degree of voluntariness or controllability in this context. No evidence was found to support an adjustment to the road accident VSL for scale of the accident (i.e. in the case of the underground accident or residents proximity to airports contexts). Thus, the limited evidence suggests context relating to voluntariness is likely to be important in determining WTP, but the weight of evidence for this is not yet strong enough to draw this as a strong conclusion, nor to adjust VSL and VLY values for air pollution exposure to account for a high degree of involuntariness.

Magnitude of Risk Change

63. A point to be observed when using the Contingent Valuation method for eliciting the willingness to pay for a reduction in probabilities of death is how sensitive the estimates are to changes in risk. Economic theory suggests that willingness to pay to reduce small probabilities of death should be

increasing with the magnitude of risk reduction, and be approximately proportional to this magnitude, assuming that risk reduction is a desired good. For example, if a reduction in annual mortality risk is valued at a certain amount of money, then a larger reduction in risk should be valued at a larger amount of money. In addition, the difference between the values should be proportional to the difference in risks, ignoring the income effect.

64. Hammitt and Graham (1999) discussed some reasons why stated willingness to pay are often not sensitive to variation in risk magnitude. One possible reason, they argued based on the review of several CVM studies, is that respondents might not understand probabilities or lack intuition for the changes in small probabilities of death risk. Another possibility relates to the fact that respondents might not treat the given probabilities as given to them. As a consequence, stated willingness to pay would not be proportional to the amount of risk reduction given to respondents, but should be proportional to changes in perceived risk.

65. In order to test for this, an internal test of sensitivity to magnitude, within a given sample, can be performed, where the respondent is asked for willingness to pay for different changes in risk in the same questionnaire. An 'external' test of sensitivity to magnitude occurs when different samples are used to compare the willingness to pay estimates, i.e. different respondents are asked about their willingness to pay for different risk reductions and there is no possibility of co-ordinating their responses. Internal tests are more likely to be successful because respondents are likely to base their responses to willingness to pay questions about one risk reduction on their answers to previous questions about a different risk change, anchoring their answers on their previous responses and enforcing some degree of consistency. Alberini *et al*. (2001) find that WTP for risk reductions varies significantly with the size of the reduction in the Canadian application of the present survey instrument. Mean WTP for an annual reduction in risk of death of 5 in 10,000 in this case was about 1.6 times WTP for an annual risk reduction of 1 in 10,000, showing sensitivity to the size of the risk reduction, but not strict proportionality.

Value of Children Versus Adults and Altruism

66. OECD (2004) reviews the evidence on economic valuation of mortality among children, and concludes that children have neither the cognitive capacities nor financial resources to state reliable preferences in SP surveys. Thus, society's perspective is the best perspective from a policy point of view, but it is not applied to children's preferences due to the difficulty in distinguishing between paternalistic and non-paternalistic altruism and, thus, the problem of double-counting due to altruism. With paternalistic altruism it would be appropriate to add up WTP across individuals. Therefore parents are asked about the value they attribute to their children's mortality risk. Some studies find the values of children's health benefits being higher than those of adults, while others find the two values to be similar, and one study found the value to be less.

5. Methodological Difficulties and Uncertainties

67. Methodological difficulties and uncertainties of HW and SP studies have been discussed in sections 3 and 4. Overall, Stated Preference methods are recommended for both mortality and morbidity valuation due to their great advantages and flexibility.

68. Based on the review in the preceding two sections, we can identify the following critical issues in morbidity and mortality valuation, which all need further research.

1. Non-linearity of value of impacts
    a. How does value of a symptom day vary with number of symptom days?
    b. How does VSL and VLY vary with age?

    c. Context–specific values (including cause of death/morbidity)
    d. Impact of health status on VLY

2. Time, discounting and latency

    a. Social vs. individual discount rate in the case of latency

3. Aggregation and equity

    a. Same value for health impacts in countries with different income levels?
    b. Same value for children and adults?
    c. Risk of double-counting due to altruism when parents asked to value children?
    d. How to take cultural and/or institutional differences into account

4. Risk perception

    a. Can people distinguish between small differences in mortality risks (scope tests)?
    b. How to value risk of contracting chronic illnesses?

69. Interim recommendation for values and how to treat these critical issues are outlined in section 6.

6. Implications for Policy Makers

Interim Recommendations of Ranges for Mortality and Morbidity

70. Due to time and resource constraints, most cost-benefit analyses (CBAs) of environmenmtal policy inaction versus action rely on value transfer. In spite of the increased uncertainty this procedure introduces, compared to conducting new health valuation surveys for each project analyzed, this is the best available approach to incorporate health values in CBAs (and would be particularly helpful in cases where costs and benefits are far apart).

71. Values should be based on Stated Preference Approaches, and be close to the EC DG Environment recommended interim VSL of EUR 1 million for acute mortality impacts from air pollution (See Appendix 2), which takes into account a 0.7 adjustment for age 70+ (but evidence for impact of age on VSL is still mixed). These values also agree well with the VSL estimates shown in Table 2. They are however, lower than those generally used by the US EPA (see Appendix 2).

72. No adjustment is made for health status, but a central discount rate of 4% is used for latency (which is lower than the private discount rates observed). However, Krupnick (2004) argues that VSL should be higher, not ignoring the higher values from the 1 in 10,000 annual risk reduction. He also points out that VSLs appear to be different across countries, and on efficency grounds different VSLs should be used. The question of whether such distinctions should be made in policy decisions is necessarily a political decision automatically.

73. The use of VLYs differentiated by age and health status could be appropriate for valuing mortality impacts from air pollution, but we need more studies estimating VLY directly (instead of computing it from VSL). For air pollution, mortality values for diseased health states should be used.

74. New results from ongoing CV surveys in the EU New Member States, applying the same CV instrument as reported in Table 2, should also be incorporated. New valuation studies of both mortality and

morbidity in developing countries are also needed, to compare estimates and assess whether the same values should be used in all countries from an ethical point of view.

75. For the valuation of morbidity endpoints from exposure-response for air pollution, the results from the 5-country study (Ready *et al*., 2004a) should be used; combined with new estimates of respiratory hospital admissions and days of breathing discomfort from Chilton *et al*. (2004).

76. For chronic morbidity, the estimate used in CAFE (Holland *et al*., 2004) of EUR 200,000 per case is based on US studies, but the purchasing power parity (PPP) rather than the prevailing exchange rate should be used. The literature underlying this value is unusually thin, consisting of only two studies that are comparable related through the use of the same methodology: Viscusi *et al*. (1991) and Krupnick and Cropper (1992).

Social Cost of Policy Inaction

77. In a cost-benefit analysis of measures to reduce pollution to air, water and soil, the aggregate costs of policy inaction are needed. For environmental health costs, we need to be able to estimate the number of different ill health episode/cases, ranging from Minor Restricted Activity Days (due to coughing) to premature death/loss of life expectancy, and multiply by transferred economic unit values for each episode.

78. A recent CV study of measures to reduce air pollution in Poland (Dziegielewska and Mendelsohn, 2005) showed health impacts to make up 61-77% of total costs of air pollution inaction (which also included reduced visibility, and costs to materials, cultural heritage and ecosystems). Taking into account that this study covered more damage components than US EPA’s (1999) aggregation exercise using transferred unit values, where total health costs made up 96%, these two approaches both show health costs to dominate the total cost of environmental policy inaction. However, interestingly, a larger proportion of the health costs were morbidity impacts (26-28% of the total value) in this original CV study, compared to morbidity (6%) and mortality (90%) from US EPA’s aggregation and value transfer exercise. Dziegielewska and Mendelsohn (2005) also found that the costs of air pollution policy inaction compared to a 25-50% reduction in premature deaths from reduced air pollution, was in the order of 0.8-1.0% of Gross Domestic Product (GDP).

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APPENDIX 1: Review of labour market (Hedonic Wage) and Contingent Valuation studies

VALUE OF STATISTICAL LIFE (VSL) ESTIMATES (Mean values in millions 1997 US dollars)		
Study	**Method**	**Value of Statistical Life**
Kneisner and Leeth (1991 - US)	Labor Market	$0.7
Smith and Gilbert (1984)	Labor Market	$0.8
Dillingham (1985)	Labor Market	$1.1
Butler (1983)	Labor Market	$1.3
Miller and Guria (1991)	Contingent Valuation	$1.5
Moore and Viscusi (1988)	Labor Market	$3.0
Viscusi, Magat and Huber (1991)	Contingent Valuation	$3.3
Marin and Psacharopoulos (1982)	Labor Market	$3.4
Gegax *et al*. (1985)	Contingent Valuation	$4.0
Kneisner and Leeth (1991 - Australia)	Labor Market	$4.0
Gerking, de Haan and Schulze (1988)	Contingent Valuation	$4.1
Cousineau, Lecroix and Girard (1988)	Labor Market	$4.4
Jones-Lee (1989)	Contingent Valuation	$4.6
Dillingham (1985)	Labor Market	$4.7
Viscusi (1978, 1979)	Labor Market	$5.0
R.S. Smith (1976)	Labor Market	$5.6
V.K. Smith (1976)	Labor Market	$5.7
Olson (1981)	Labor Market	$6.3
Viscusi (1981)	Labor Market	$7.9
RS.Smith (1974)	Labor Market	$8.7
Moore and Viscusi (1988)	Labor Market	$8.8
Kneisner and Leeth (1991 - Japan)	Labor Market	$9.2
Herzog and Schlottman (1987)	Labor Market	$11.0
Leigh and Folson (1984)	Labor Market	$11.7
Leigh (1987)	Labor Market	$12.6
Garen (1988)	Labor Market	$16.3
From *Guidelines for Preparing Economic Analyses, US EPA 2000.* Derived from EPA (1997); see Viscusi (1992) for complete references for each study.		

APPENDIX 2: Comparing Common Practice of DG Environment of the European Commission and US Environmental Protection Agency

Issue	DG Environment	US EPA*
Conceptual Approach	VSL, adjusted	VSL, adjusted
Base Estimate	Best Estimate from UK CVM transport studies Central value: €1.4 million[13]	Collection of studies, mostly hedonic wage Central value: $6.1 million
Sensitivity around Base Estimate	Upper limit: €3.5m from ExternE Lower estimate: €0.65m (requiring fewer adjustments)	Weibull distribution fitted to collection of study means
Age	0.7 adjustment for age 70+	No adjustment Alternative calculations with VLY and age-specific VOSL
Cross-Sectional Income Differences	No adjustment, EU15 PPP adjustment between EU15 and Accession Countries	No adjustment
Growth in Real Income over Time	See below	Adjust for changes in per capita GDP Central income elasticity of WTP of 0.4
Latency	Discount over latency period at 4 percent real discount rate Sensitivity rate of 2 percent to reflect likely rise in real income over time	Has varied: No discounting Discount over latency period at 2 to 3 percent real discount rate
Cancer Premium	+50% adjustment	No adjustment
Health Status	No adjustment	No adjustment

Note: * This reflects recent practices at EPA, but not in all cases the current EPA practice.

Source: Dockins C. and S. White 2005: Benefit Transfer for Estimating the Value Reduced Premature Mortality Risks: Practice on both sides of the Atlantic; Chapter 5 in Navrud, S and R. Ready (eds.) 2005: *Environmental Value Transfer: Issues and Methods*. Springer (Kluwer Academic Publishers), Dordrect, The Netherlands. Forthcoming.

[13] The exchange rate used is: 1 US $= 0.93 euro (in 2002).

APPENDIX 3: Exposure-Response Functions recommended by the Externe Project Series

Receptor	Impact	Reference	Pollutant[1]	f_{er} [1]
ASTHMATICS				
Adults	Bronchodilator Usage	Dusseldorp *et al.*, 1995	PM_{10} $PM_{2.5}$	0.163 0.272
	Cough	Dusseldorp *et al.*, 1995	PM_{10} $PM_{2.5}$	0.168 0.280
	Lower respiratory symptoms (wheeze)	Dusseldorp *et al.*, 1995	PM_{10} $PM_{2.5}$	0.061 0.101
Children	Bronchodilator usage	Roemer *et al.*, 1993	PM_{10} $PM_{2.5}$	0.078 0.129
	Cough	Pope and Dockery, 1992	PM_{10} $PM_{2.5}$	0.133 0.223
	Lower respiratory symptoms (wheeze)	Roemer *et al.*, 1993	PM_{10} $PM_{2.5}$	0.103 0.172
All	Asthma attacks (AA)	Whittemore & Korn, 1980	O_3	4.29 E-03
ELDERLY 65+				
	Congestive heart failure	Schwartz and Morris, 1995	PM_{10} $PM_{2.5}$ CO	1.85 E-05 3.09 E-05 5.55 E-07
CHILDREN				
	Chronic cough	Dockery *et al.*, 1989	PM_{10} $PM_{2.5}$	2.07 E-03 3.46 E-03
ADULTS				
	Restricted activity days (RAD)[2]	Ostro, 1987	PM_{10} $PM_{2.5}$	0.025 0.042
	Minor RAD [3]	Ostro and Rothschild, 1989	O_3	9.76 E-03
	Chronic bronchitis	Abbey *et al.*, 1995 (after scaling)	PM_{10} $PM_{2.5}$	2.45 E-05 3.90 E-05
ENTIRE POPULATION				
	Respiratory hospital admissions (RHA)	Dab *et al.*, 1996	PM_{10} $PM_{2.5}$	2.07 E-06 3.46 E-06
		Ponce de Leon, 1996	SO_2 O_3	2.04 E-06 3.54 E-06
	Cerebrovascular hospital admissions	Wordley *et al.*, 1997	PM_{10} $PM_{2.5}$	5.04 E-06 8.42 E-06
	Symptom days	Krupnick *et al.*,	O_3	0.033
	Cancer risk estimates	Pilkington *et al.*, 1997; based on US EPA	Benzene 1,3-butadiene	1.14 E-07 4.29 E-06
	Acute Mortality (AM)	Spix *et al.* / Verhoeff et al, 1996	PM_{10} $PM_{2.5}$	0.040% 0.068%
		Anderson *et al.* / Touloumi *et al.*, 1996	SO_2	0.072%
		Sunyer *et al.*, 1996	O_3	0.059%

Notes:

[1] The exposure response slope, f_{er} , has units of case events per year per person per µg/m^3, except for mortality which is expressed as percentage increase per µg/m^3 Sources: (European Commission, 1995 Hurley *et al*.., 2000). Within ExternE, sulphates are treated as $PM_{2.5}$ and nitrates as PM_{10}.

[2] Assume that all days in hospital for respiratory admissions (RHA), congestive heart failure (CHF) and cerebrovascular conditions (CVA) are also restricted activity days (RAD). Also assume that the average stay for each is 10, 7 and 45 days respectively. Thus, net RAD = RAD - (RHA*10) - (CHF*7) - (CVA*45).

[3] Assume asthma attacks (AA) are also minor restricted activity days (MRAD), and that 3.5% of the adult population (80% of the total population) are asthmatic. Thus, net MRAD = MRAD - (AA*0.8*0.035). Source: European Commission (1999)