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A REVIEW OF RECENT POLICY-RELEVANT FINDINGS FROM THE ENVIRONMENTAL HEALTH LITERATURE

This review of recent policy-relevant findings from the environmental health literature was prepared by Alistair Hunt and Julia Ferguson, University of Bath, UK. The focus of the paper is on environmentally related morbidity impacts – not on mortality impacts.

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FOREWORD

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The paper is issued under the responsibility of the Secretary-General of the OECD.

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A REVIEW OF RECENT POLICY-RELEVANT FINDINGS FROM THE ENVIRONMENTAL HEALTH LITERATURE

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Part I: Introduction

1. This paper, commissioned by the OECD Environment Directorate, provides a broad overview of the monetary valuation of environmental health risks, with a focus on non-fatal, or morbidity, health impacts. It may therefore be regarded as complementary to the recent OECD review on mortality valuation. More particularly, the paper aims to address a number of questions that have been posed, and serve to inform a closer marriage between the interests of the research community, who generate health value estimates, with the needs of the policy communities who wish to use them.

2. In attempting to meet this objective, the most important questions that should be addressed include the following: What are the most important environmental health impacts? What are the policy appraisal needs? To what extent are these needs currently being met? An important subsidiary question is then: If current policy needs are not being met, why is that the case, and how can we ensure that they are met in the future? These are empirical questions and, consequently, in reviewing the evidence the paper focuses on the availability, robustness and up-take of monetary values of health endpoints in policy contexts. However, their robustness at least relies on unit values being derived from empirical work that has a strong theoretical basis. The generic theory is therefore outlined in Part II, below. The scope of the review is limited by a focus on health valuation experience in the last 15 years and, primarily, in OECD countries. However, the latter constraint is relaxed when considering issues associated with trans-boundary pollution and, in particular, climate change impacts.

3. The paper has the following structure. Part I provides an overview of the ways in which monetary values for environmental health impacts have been, or may in the future be, used in decision-making related to public policy. The focus on public policy therefore implies that valuation is viewed here from a social welfare perspective. Part II then summarises the current health unit value estimates in relation to their availability for use in such decision processes. At this point we also highlight the principal methodological issues that remain unresolved in primary study value derivation, and that may determine their adequacy in such uses. Part III complements this by investigating the adequacy of the current practice of transferring existing values to a different (temporal or geographical) policy context than from which they were originally derived: benefit – or value – transfer. Finally, Part IV summarises best practice in morbidity valuation derivation and application and provides some pointers for future research priorities in relation to current gaps, emerging environmental health issues, and outstanding methodological issues.

Part I: Policy Needs

4. In broad terms, the rationale for policy arises as a result of the public policy requirement to internalise externalities, of which the health effects of environmental pollution are examples. Before exploring more precisely the ways in which public policy decision-makers may be expected to incorporate information on externalities, it is worth re-stating the possible sources and forms of environmental health externalities. The chapter on Cost of Policy Inaction in OECD (2008) describes the health effects associated with air pollution and water pollution in some detail. These effects are summarised in Table 1 and Table 2, whilst Tables 3-6, summarise a number of environmental health impacts that result from other environmental media or processes such as soil, radioactivity and climate change.

5. A constraint on the efficacy of some forms of policy analysis, such as cost-benefit analysis, is the extent to which the physical impacts can be quantified.[1] In the context of health impacts, quantitative epidemiological relationships that link pollution exposure to specific health outcomes are required. If these relationships – known as exposure-response functions – exist, and it is possible to model the change in the number of cases of individual health impacts as a result of a policy intervention, then the monetary value per case can be applied. This application can then serve to express the aggregate change in population health in a common metric, and allow comparison with other costs and benefits of a given policy intervention. The health impacts listed in tables 1-6 all have some quantitative evidence that potentially allow such a policy appraisal to be undertaken. These lists are not comprehensive though they attempt to capture the majority of impacts quantified to date. These tables also indicate whether monetary valuation of each health impact has been attempted. Specifically, valuation is indicated to have been undertaken where all three welfare components – medical treatment costs, opportunity costs of lost productivity and dis-utility from pain and suffering, described further in part II of this paper – are known to have been estimated in empirical studies. Populations affected are indicated by "Adult" and "Child" notations where separate pollution exposure – response functions are specified; Additionally, Old = elderly aged 65 and over, whilst Asth = Asthmatics.

6. Table 1 shows that, of the health impacts quantified, all but broncho-dilator usage have monetary values existing that have been or could be used in policy appraisal. The extent of this coverage perhaps reflects the greater degree of attention that has been given to the regulation of air pollution relative to other pollutants, at least based on economic, rather than precautionary, criteria.

Table 1. Summary of Health Impacts from Outdoor Air Pollution that are Quantified and/or Monetised

Outdoor air pollution	
Health Impact	**Monetary valuation – data exists**
Premature mortality (chronic)	Yes
Premature mortality (acute)	Yes
Respiratory hospital admission	Yes
Cerebro-vascular hospital admission	Yes
Cancer (lung) (fatal/non-fatal)	Yes
Chronic bronchitis (Adult)	Yes
Restricted activity days (RADs) (Adult)	Yes
Minor RADs (Adult)	Yes
Chronic cough Child	Yes
Congestive heart failure Child	Yes
Asthma attacks Asth	Yes
Lower respiratory symptoms	Yes
Cough Child; Asth	Yes
Bronchodilator usage Asth:	No

[1] Of course, policy is also made in situations where quantitative data is lacking, *e.g.* on the basis of the precautionary principle.

7. The total coverage, in terms of monetary valuation, of the health impacts from indoor air pollution primarily reflects the possibility of transferring values from the impacts common to outdoor air pollution. The extent of the coverage, however, belies the fact that the range of values for each end-point is generally limited to a small number of studies. Atopy, for example, has – to our knowledge – only had one valuation study undertaken on it; this was carried out in Japan, (Toshiya, 2001).

Table 2. Summary of Health Impacts from Indoor Air Pollution that are Quantified and/or Monetised

Indoor air pollution	
Health Impact	**Monetary valuation – data exists**
Lung Cancer	Yes
Leukaemia	Yes
Respiratory hospital admission	Yes
Cardiovascular hospital admissions	Yes
Respiratory - Symptoms: those with asthma (Asthmatics)	Yes
Restricted activity days (RADs) (Adult)	Yes
Lower respiratory symptoms	Yes
Atopy, conjunctival irritation, Allergy/Irritant	Yes
Asthma attacks (Asthmatics)	Yes
Neuro-devt. disorders (Child)	Yes

8. The near-comprehensive coverage of monetisation for air pollution contrasts starkly with the coverage for water pollution in Table 3, and for the health impacts listed in Tables 4 - 6. The relative ease with which air pollutant emissions can be modelled may be the main reason for the lack of monetisation. However, it is also likely to be the case that cost-benefit analysis has only recently been considered as an appraisal method in the regulation of activities resulting in these emissions. This factor is likely to be exacerbated by the fact that these health impacts have – in the main - only recently been identified and quantified.

Table 3. Summary of Health Impacts from Water Pollution that are Quantified and/or Monetised

Water	
Health Impact	**Monetary valuation – data exists**
Gastro-intestinal disorders, incl. diarrhoea	Yes
Dental fluorosis	No
Blue baby syndrome	No
Skin lesions	No
Infections: eye, ear, skin	No
Cholera	No
E. Choli	No
Respiratory disease	Yes

9. To date, almost all monetisation research in the context of noise has been interested in annoyance, which has been assumed to act as a catch-all for other health impacts associated with noise. This reflects the coverage indicated in Table 4. However, it is now recognised that annoyance impacts may be identified separately from the health impacts, and that the latter impacts may be significant, if not hitherto acknowledged.

Table 4. Summary of Health Impacts from Noise Pollution that are Quantified and/or Monetised

Noise	
Health Impact	**Monetary valuation – data exists**
Ischemic heart disease / myocardial infarction	Yes
Hypertension	No
Cognitive impairment	No
Hearing impairment	No
Annoyance	Yes

10. As listed in Table 5, the health impacts of heavy metals, such as arsenic, lead, cadmium and others, are separated from those associated with water, given the more complex pathways associated with many of them. The table indicates that cancer impacts have had some attention in valuation terms. Neuro-developmental disorders have also been subject to valuation as a result of recent regulatory efforts in relation to lead poisoning.

Table 5. Summary of Health Impacts from Heavy Metals & Radioactivity that are Quantified and/or Monetised

Heavy Metals & Radio-activity	
Health Impact	**Monetary valuation – data exists**
Lung Cancer	Yes
Skin cancer	Yes
Leukaemia	Yes
Osteoporosis (Old)	No
Renal dysfunction	No
Anaemia	No
Neuro-devt. disorders, (Child)	Yes
Other cancers	Yes

11. Table 6 highlights a number of health impacts associated with climate change globally. These have only recently been recognised and consequently have yet to enter appraisal practices in relation – for example – to adaptation decision-making.

Table 6. Summary of Health Impacts from Climate Change that are Quantified and/or Monetised

Climate Change	
Health Impact	**Monetary valuation – data exists**
Premature mortality	Yes
Cardiovascular illness	Yes
Cardio-pulmonary disease	No
Lymes Disease	No
Malaria	No
Diarrhoea	Yes
Injuries	Yes

12. One observation to make at this point – relevant to the discussion of valuation methods, below – is that a number of the health impacts listed above are associated with different environmental pollutants and media, thereby raising the issue for valuation as to whether the source of pollution is important in determining the welfare change to be measured in an empirical valuation study. For example, whether the source is important may influence whether this information is provided in a stated preference study; it may also determine whether the results from a study where the source is known can be transferred to another context with a different source.

13. The following paragraphs offer an overview of the empirical studies of morbidity valuation. The health end-points for which studies have been undertaken are listed, after which the body of empirical evidence is evaluated against criteria relating to the quantity and quality of studies, the extent to which cost

of illness data exists, and a broad indication of the significance of each health end-point relative to others in existing RIAs.

14. An assessment of the *quantity* of the dis-utility WTP studies allows us to identify whether there is a narrow or wide body of evidence and therefore whether it is possible to establish whether the findings of one or more studies can be corroborated easily. The *quality* criterion is also related to the transferability of the available evidence since it is likely that if a study has been undertaken more than a few years ago, the methodology will now be considered to be outdated.

Table 7. Rating: WTP studies -- Quantity and Quality

	Low	Medium	High
Quantity	≤ 2 studies worldwide	2 – 3 studies worldwide	≥ 4 studies worldwide
Quality	Clear serious methodological weaknesses and/or low sample size (SP) < 200	Some methodological weaknesses and/or sample size 200-500	State-of-art method and sample size > 500

15. *Cost of Illness (COI) Data.* Existence of this data allows us to estimate total social welfare costs of the health endpoint by aggregating loss in utility, and the COI. The criterion is set out in Table 8.

Table 8. Rating of COI Evidence Base

	Low	Medium	High
COI Data	≤ 2 studies worldwide; no EU studies in past 5 years	2 – 3 studies worldwide; 1 – 2 EU studies in past 5 years	≥ 4 studies worldwide; ≥ 2 EU studies in past 5 years

16. *Significant total costs within HIAs*. A further important criterion in the evaluation of the empirical evidence is the extent to which the aggregate costs of the selected health endpoint are thought to make up a significant portion of the external costs estimate. It should be noted, moreover, that the unit cost need not be large if the frequency of the health endpoint and/or the number of people affected is large. The criterion is laid out in Table 9.

Table 9. Rating of Significance of end-point in total HIA costs

	Low	Medium	High
Significant total costs	< 2% of impact costs of multi-pollutant HIA	2-10% of impact costs of multi-pollutant HIA	> 10% of impact costs of multi-pollutant HIA

17. The criteria are then applied to the evidence; using our own judgement we evaluate how the empirical valuation literature relating to each health end-point is rated. Our (admittedly subjective, and therefore indicative only,) evaluation is then summarised in Table 10, below. It is clear from this evaluation that the scores against the quality and quantity criteria are either Low or Medium; there are no High ratings given. Cancers and health end-points associated with complex pollutants are notably low-rating. It is also evident that the cost of illness data is also limited for the vast majority of health end-points considered.

18. The use of health, as well as other, externality values has principally been as inputs to the determination of a) the economic efficiency of projects or policies, *i.e.* in cost-benefit analysis; b) environmental costing, *i.e.* externality internalisation in policy design; c) environmental accounting, *i.e.* inclusion of non-market impacts in national or corporate accounting exercises, and; d) levels of compensation, *i.e.* where legal requirements demand that individual polluters compensate welfare losses of identified victims. Of these decision-making aids, cost-benefit analysis – now seen as integral to the geographically widespread legal requirement of undertaking regulatory impact assessments for new public policies or projects – is by far the most common vehicle for use of health valuation.

Table 10. Summary of environmental health valuation evidence

	Quantity	Quality	COI Data	Signif. Ext costs
Premature mortality (chronic)	M	M	M	H
Premature mortality (acute)	M	M	M	M
Respiratory hospital admission	M	M	H	H
Cerebro-vascular hospital admission	M	M	H	H
Cancer (lung) (fatal/non-fatal)	L	L	M	H
Chronic bronchitis	L	L	M	H
Restricted activity days	M	M	H	H
Minor RADs	M	M	M	M
Chronic cough	M	M	M	M
Congestive heart failure	M	M	M	M
Asthma attacks	M	M	M	M
Lower respiratory symptoms	M	M	M	M
Cough	M	M	M	M
Bronchodilator usage	L	L	M	M
Atopy, conjunctival irritation, Allergy/Irritant	L	L	L	L
Ischemic heart disease / myocardial infarction	M	M	H	M
Hypertension	M	M	H	M
Cognitive impairment	L	L	M	L
Hearing impairment	L	L	M	L
Skin cancer	L	L	M	H
Leukaemia	L	L	M	M
Osteoporosis	L	L	M	L
Renal dysfunction	L	L	M	L
Anaemia	L	L	M	L
Neuro-devt. disorders	L	L	M	M

19. Use of externality values may also be viewed in terms of their different roles in the wider decision-making process. For example, methodological analysis by Gerdes *et al*. (2008) identifies four broad categories of use: instrumental; conceptual; political, and; symbolic. Thus, health values as used in the determination of the national emission ceilings for emissions of specific air pollutants in the European Union, (EU), may be regarded as having direct/instrumental use since they are supposed to bring rationality to policy-making, initiate action, and directly improve policies. Conceptual use or 'enlightenment*'* is defined as "the percolation of new information, ideas and perspectives into the arenas in which decisions are made" (Weiss, 1999, p. 471). This represents awareness-raising, forming of opinions, and the identification of policy instruments. National environmental accounting exercises can be regarded as an example of this type of use.

20. The other two types of use are perhaps more cynical in flavour. Political use is principally about legitimisation, in which research is a rationalistic ritual aimed at justifying decisions that have already been taken or policies that are already in place (Weiss, 1999, p. 477), or tactical, as when research is initiated to postpone decision-making by referring to an ongoing study (Vedung 2001). Closely related to this, symbolic use exists where research is initiated or maintained to give the impression of a rational process though, in reality, decisions are motivated by other factors, *e.g.* short-term economic interests, un-related to the research or its results. In many cases, the determination of compensation-levels may be subject to these latter two uses of health, and other, externality valuation.

21. Whilst the four decision-making uses have been previously discussed in terms of the different degrees of accuracy in the values required for each, (see *e.g.* Navrud and Pruckner (1997)), it is also likely that the use of individual values will be more heavily contested – and therefore higher degrees of accuracy expected – depending on the role, or roles, their use is playing. For example, methodological uncertainties or variation in values from different studies may be exploited for the subsequent justification of political or symbolic uses. These factors are likely to be important when considering trade-offs between conducting

primary studies and value transfer, and the specification of these, as discussed below. A priori, it is to be expected that valuations that inform compensation are likely to be most contested and therefore most needful of reduced uncertainties. Whilst cost-benefit analysis and environmental costing are also contested, though perhaps to a lesser degree, environmental accounting – with its present use being primarily informational and awareness raising – is least constrained by uncertainties in its derivation.

22. Table 11 gives an impression of the extent and breadth of applications of environmental health impact estimation undertaken in recent years across a range of countries world-wide. The table identifies the general purpose of the individual studies listed. The list of studies is not comprehensive but does provide an overview of a number of the principal studies undertaken since the mid-1990s. Those studies that are un-shaded report an application within a Regulatory Impact Analysis (RIA). Those studies that are shaded report either a scoping study in a region or country – generated either by the researchers themselves or by a public agency – or an *ex-post* evaluation of a regulation. Details of the pollutants and associated health endpoints that are considered in each study are given. The majority of the studies are concerned with the classical air pollutants: particulates, sulphur dioxide, nitrogen oxide, carbon monoxide, as well as low-level ozone. Benzene, Lead, Volatile Organic Compounds and some Heavy Metals also feature. The sources of the pollutants are predominantly energy and transport sector-related, with a handful of studies focusing on industrial sectors. Studies supported by the USEPA and the European Commission are the most numerous. Indeed, the common methodological underpinnings that the majority of these studies share are derived from a joint US-EU research collaboration on the social costs of fuel cycles, undertaken in the early 1990s.

23. Apart from the study, Office of Air Quality Planning and Standards, USEPA (2002), which includes relatively under-researched pollutants such as formaldehyde, the results of the studies are presented in both quantitative and monetised terms. In addition, all of the studies listed except that by Monzon *et al.* (2004), which includes resource costs only, adopt a willingness-to-pay (WTP) measure of the welfare costs. The health endpoints considered by the studies are, of course, limited by the availability of epidemiological exposure-response functions that allow quantification of the relationship between human exposure to a pollutant and a consequent health effect. These functions tend to be transferred between contexts and so generate a group of health endpoints common to a large number of the studies. In a similar way, the unit values used to monetise these health impacts have also been – in the majority of studies cited – transferred from the original geographical study site.

24. Table 12 documents a sample of studies that feature policy-related uses of environmental health valuation in the context of water, waste, noise and land-based pollutants. Clearly, they are not as numerous as those relating to air pollution; however, the studies that have been undertaken appear to have served a similar range of purposes, including providing inputs to RIAs or scoping out total costs of policy inaction. The range of health impacts also varies over the different environmental media considered, and differs again from those associated with air pollution.

25. As highlighted earlier, the majority of the studies listed in Table 11 use health valuation as an input to a cost-benefit analysis within an RIA. However, some studies present health impact costs in the context of estimating either costs of policy inaction or welfare damage costs within a Green Accounting exercise, where environmental welfare costs are compared with other aggregate welfare measures, such as Gross Domestic Product, that exclude these costs. An example of an assessment of the measurement of environmental health costs in a Green Accounting context is given in Table 13, from Droste-Franke and Friedrich (2004). The table presents welfare damage costs across EU15 countries resulting from emissions to air of particulates (PM_{10}), sulphur dioxide and ozone from within these countries, and including trans-boundary pollution. In addition, the unit values adopted in the generation of these results are also presented.

Table 11. Overview over Recent Environment-related Health Impact Valuation Studies

Author(s) / Date	***Pollutant***	***Policy input / Purpose***	***Health Measurement***
North America			
Abt Associates Inc – 2002	PM	Annual benefits associated with South Appalachian Mountain air pollution Controls, USA	Mortality; CB; AB.
Austin, D *et al.*– 1998	SO_2, NO_x, PM	Environmental Benefits from air pollution emissions reduction to 2010, Maryland, USA	Acute mortality; Chronic Mortality; ERV; RHA; RAD; MRAD; Lower respiratory symptoms; Asthma attacks
Bloyd, C – 2002	NO_x, SO_2	Investment in Electricity Transmission and Ancillary Environmental Benefits in Western States, USA	Acute mortality; Chronic Mortality; ERV; RHA; RAD; MRAD; Lower respiratory symptoms; Asthma attacks
Burtraw,D *et al.* – 1997	SO_2 and NO_x	Costs and Benefits of Reducing Acid Rain (USA)	Acute mortality; Chronic Mortality; ERV; RHA; RAD; MRAD; Lower respiratory symptoms; Asthma attacks
Burtraw and Toman, – 1997	SO_2, NO_x, PM	Benefits of Reduced Air Pollutants from Greenhouse Gas Mitigation Policies (USA)	Acute mortality; Chronic Mortality; ERV; RHA; RAD; MRAD; Lower respiratory symptoms; Asthma attacks
Burtraw and Toman – 2000	SO_2, NO_x, CO	Ancillary Benefits of Greenhouse Gas Mitigation Policies	Acute mortality; Chronic Mortality; ERV; RHA; RAD; MRAD; Days of bronchodilator usage; Cough days; Lower respiratory symptoms; Asthma attacks; Cancer
Burtraw, D *et al.* – 2001	NO_x	Health Benefits (Relating to Carbon Tax) Ancillary Benefits of Reduced Air Pollution in the United States from Moderate Greenhouse Gas Mitigation Policies in the Electricity Sector	Acute mortality; Chronic Mortality; ERV; RHA; RAD; MRAD; Lower respiratory symptoms; Asthma attacks;
Burtraw, D *et al.* – 2001	NO_x	NO_x Reduction Policies from Electricity Generation	Acute mortality; Chronic Mortality; ERV; RHA; RAD; MRAD; Lower respiratory symptoms; Asthma attacks
Chestnut and Mills – 2005	SO_2; NO_x	Benefits and Costs of the US Acid Rain program	Acute mortality; Chronic Mortality; ERV; RHA; CHA; CHF; RAD; MRAD; Upper and Lower respiratory symptoms; Work Loss Days
Kahn, ME – 1996	O_3; CO	New estimates of the benefits of vehicle emission regulation, Chicago, USA	RAD
Levy *et al.* – 2001	O_3	Health Benefits from reducing Ozone in Houston, Texas, US	Acute Mortality, RHA, MRAD; Chronic Asthma
McCubbin and Delucchi – 1999	PM, CO, NO_2, Ozone	The Health Costs of Air Pollution in the Greater Sydney Metropolitan Region, Los Angeles, USA	Acute mortality; Chronic Mortality; ERV; RHA; RAD; MRAD; Days of bronchodilator usage; Cough days; Lower respiratory symptoms; Asthma attacks; Cancer
Muller and Mendelsohn – 2007	PM; NH_3; NO_2; SO_2; VOCs	Gross annual damages due to air pollution in US	Acute mortality; Chronic Mortality
Office of Air Quality Planning and Standards USEPA – 2000	SO_2	Benefits Assessment of SO_2 reductions in 9 Western States in USA.	Acute mortality; Chronic Mortality; RAD; MRAD; Asthma attacks
Office of Air Quality Planning and Standards USEPA – 2002	VOC, CO, Formaldehyde, Methanol, etc	Regulatory Impact Analysis of the Proposed Plywood and Composite Wood Products NESHAP	Cancer (not quantitative)

Rittmaster – 2004	PM	Economic Analysis of the Human Health Effects from Forest Fires, Alberta, Canada	Chronic Mortality; ERV; RHA; CHA; Bronchitis HA; RAD; MRAD; Lower respiratory symptoms;
Stieb *et al.* – 2002	SO_2	Benefits of sulphate reductions in Toronto, Canada	RHA; CHA; ERV; RAD; Asthma Attack; Respiratory Symptom Days
USEPA – 1995	PM; Pb; O_3; CO; SO_2; NO_2	Benefits and Costs of The Clean Air Act, 1970-1990	Acute mortality; Chronic Mortality; CB; CHA; RHA; RAD; Asthma Attacks; Lower respiratory symptoms; IQ etc
USEPA – 1995	SO_2	Health Benefits from SO_2 Reductions – 1990 Clean Air Act Amendments	Acute mortality; Chronic Mortality; ERV; RHA; RAD; MRAD
USEPA – 1998	Various	Full-Cost Accounting for Decision-Making at Ontario Hydro," in "The Green Bottom Line	Acute mortality; Chronic Mortality; RHA; Cancer
USEPA – 1999		Benefits and Costs of The Clean Air Act, 1990-2010	Acute mortality; Chronic Mortality; AB; CB; ERV; RHA; RAD; MRAD; Upper and Lower respiratory symptoms; Cough days; Chronic Asthma; IQ; Asthma attacks; Work Loss Days
Europe			
AEA Technology – 1999	NO_X, SO_2, O_3	Proposals for Emission Ceilings for Atmospheric Pollutants (EU15),	Acute mortality; Chronic Mortality; CB; ERV; RHA; RAD; MRAD;
AEA Technology – 2001b	CO and Benzene	Economic Evaluation of Air Quality Targets for CO and Benzene	Mortality; Congestive heart failure; Cancer
AEA Technology – 2001a	PAH's	Economic Evaluation of Air Quality Targets for PAH's (EU15)	Fatal and Non fatal Cancer
AEA Technology – 2006	SO_2, NO_x, PM and VOC	Review of UK Air Quality Strategy	Acute mortality; Chronic Mortality; RHA; CHA
AEAT – 1998	Ozone (NO_x and O_3)	Economic Evaluation of Air Quality Targets for Tropospheric Ozone (EU15)	Acute mortality; Chronic Mortality; RAD; CB.
Commission of the European Communities – 2005	PM; O_3, NO_x, VOC; NH_3	Impact Assessment on the Thematic Strategy on Air Pollution – Clean Air for Europe (CAFÉ) Cost-Benefit Analysis	Acute mortality; Chronic Mortality; CB; ERV; RHA; CHA; RAD; MRAD; Days of bronchodilator usage; Cough days; Symptom Days; Lower respiratory symptoms; Asthma attacks;
Department of Health – 1999	PM_{10}; SO_2; O_3	Total per annum benefits net of NHS costs relating to reductions; (UK)	Acute mortality; Chronic Mortality; RHA
European Commission DG Research – EXTERNE	PM; O_3, NO_x, VOC; NH_3	Total environmental damage cost estimation	Acute mortality; Chronic Mortality; CB; ERV; RHA; CHA; RAD; MRAD; Days of bronchodilator usage; Cough days; Symptom Days; Lower respiratory symptoms; Asthma attacks;
Entec UK Ltd – 2001	Arsenic, Cadmium, Nickel	Compliance with Heavy Metal Limit Values for EU15	Fatal Cancers
ERM Economics – 1996	Lead; SO_2	CBA for Integrated Pollution Control – Hypothetical industrial plant	IQ; Blood Pressure
Monzon, *et al.* – 2004	Not specified	Social and Health Effects of Transport-Related Air Pollution in Madrid, Spain	Acute mortality; Chronic Mortality; ERV; RHA; RAD; MRAD; *Cost-based*

Olsthoorn *et al.* – 1999	SO_2, NO_2 and PM	Cost Benefit Analysis of European Targets for SO_2, NO_2 and PM in Cities, EU15	Acute mortality; Chronic Mortality; ERV; RHA; RAD; MRAD;
Rabl – 2002	CO_2; CO; PM; N_2O; SO_2; Methane	Damage Costs of diesel vs gas buses Environmental Benefits of Natural Gas for Buses, Toulouse and Paris, France.	Acute mortality; Chronic Mortality; ERV; RHA; RAD; MRAD; Days of bronchodilator usage; Cough days; Lower respiratory symptoms; Asthma attacks; Cancer
Australia			
Department of Environment and Conservation, NSW – 2005	PM, NO_x, VOC	Health Costs of Air Pollution in the Greater Sydney Metropolitan Region	Chronic Mortality; CB; Acute Bronchitis; RHA; CHA; RAD; MRAD; Asthma attacks
Non-OECD			
Alberini *et al.*	PM, NO_2, SO_2, O_3, CO	Air pollution morbidity costs, Taiwan	Cold and non-cold – 1 or 5 days
Euston *et al.* – 2003	PM	Economic Costs of Particulate Air Pollution on Health in Singapore	Acute mortality; Chronic Mortality; CB; ERV; RHA; RAD; Cough days; Lower respiratory symptoms; Asthma attacks;
Hope and Maul – 1996	CO_2	Valuing the Impact of CO_2 Emissions, Global	Mortality
Li, J. *et al.* – 2004	SO_2; NO_x; PM	Health Benefits of Curbing Air Pollution in Shanghai",	Work Loss Days
Pearce – 1996 Costs	SO_2, NO_2, Pb, O_3, and PM	Economic Valuation and Health Damage from Air Pollution – 5 Developing Countries	Morbidity Acute mortality; Chronic Mortality; ERV; RHA; RAD; MRAD; Days of bronchodilator usage; Cough days; Lower respiratory symptoms; Asthma attacks; Cancer
Seroa da Motta *et al.*– 2000	PM, CO_2	Values for Mortality and Morbidity Cases Associated with Air Pollution in Brazil	Mortality; Respiratory Problems; Heart Failure
Strukova *et al.* – 2006	PM	Total annual cost of morbidity from PM in Ukraine	Acute mortality; Chronic Mortality; ERV; RHA; RAD; MRAD
Zhou and Tol – 2005	PM	Economic cost of Health Impacts From Particulate Air Pollution in Tianjin, China	Acute mortality; Chronic Mortality; ERV; RHA; CHA; CB; AB; RAD; Asthma Attacks

CB = Chronic Bronchitis; RHA = Respiratory Hospital Admissions; CHA = Cardiovascular Hospital Admissions ERV = Emergency Room Visits; AB = Acute Bronchitis; RAD = Restricted Activity Days; MRAD = Minor Restricted Activity Days; IQ = Intelligence Quotient

Table 12. Policy-related Applications of Health Values for Water, Waste and Noise

Authors/year/RIA-related	Pollutant	Source	Measurement
Anderson, *et al.* (2000). Yes	Bacterial; Harmful Algal blooms	General effluent	Cost of public health impacts
Blomquist, *et al.* (2003). Yes	Bacterial; Chemical	Various	Allocation of state funds to environmental programmes
Browner, *et al.* (2000). Yes	Bacterial; Chemical	Transport discharge (Truck and rail)	Benefit of final effluent guidelines (Reduction of carcinogenic risk)
Droste-Franke, *et al.* (2004)	Chemical (Heavy Metals)	Various	Fatal and non-fatal cancers (lung, leukaemia, other), IQ, Renal effects
Dwight, *et al.* (2004). No	Bacterial	Urban run-off	Cost of : Gastro-intestinal episode; Acute respiratory disease; Ear infections; Eye infections
Landrigan, *et al.* (2002). No	Chemical	Various	Costs of pediatric disease: Lead poisoning; Cancer; Asthma; Neuro-behavioural disorders
Scasny *et al.* (2008)	Chemical (Heavy Metals)	Various	Fatal and non-fatal cancers (lung, leukaemia, other), IQ, Renal effects, Osteoporosis,
Zhang, (1999). Yes	Bacterial; Chemical	Waste water	Health benefits of waste water treatment (China)
Defra – ICGB Noise Subject Group (2008). No	Noise	Various	Annual UK Impact of Noise: Heart disease; Severe sleep disturbance; Severe annoyance; Tinnitus; Learning deficit in children;
Den Boer & Schroten (2007). No	Noise	Transport (Road and Rail)	WTP for reduction in noise pollution – social costs of noise across EU22
OECD/INFRAS/Herry (2002). No	Noise	Transport (Road and Rail)	Social cost of noise in Central and Eastern Europe
EC DG Research UNITE Project (2002). No.	Noise	Transport (Road and Rail)	Myocardial Infarction; Angina Pectoris; Hypertension; Severe sleep disturbance

26. The results from the Droste-Franke and Friedrich (2004) study serve to illustrate a number of points with regard to environmental health impacts and valuation. First, the total welfare costs of environmental health impacts are given as €130 billion – equivalent to about 0.85% of total EU15 GDP for 1998. These health costs are dominant in total external costs – this study found them to constitute 90% of the total external costs of air pollution; damage to materials and crops being the other categories of damage considered. The European Commission, (European Commission, 2003), summarising the work of the ExternE research projects on energy and transport externalities, found that the marginal external costs attributable to a range of alternative energy fuel cycles were dominated by health costs. For example, for coal burnt in Germany, the total external costs were estimated to be €0.0075 per kWh, of which the health costs accounted for €0.0073 per kWh – equivalent to about 5% of the consumer price paid.

27. Table 13 shows that the health costs of air pollution are attributable to a number of mortality and morbidity endpoints. The welfare costs of premature mortality constitute the largest single cost item, at almost 68% of the total environmental health costs in Europe. Out of the 32% that are comprised of the welfare costs from fifteen different morbidity endpoints, by far the largest are the new cases of chronic bronchitis (14%), restricted activity days (9%), and symptom days (3%). This cost split across different health endpoints is confirmed in Table 14, which summarises the aggregate welfare costs in a sample of the policy application studies discussed above.

28. Whilst the results from the individual studies in Table 14 derive from a variety of policy contexts over different geographical regions and so cannot be compared against each other, a within-study comparison shows that – of the studies that quantify chronic bronchitis – this endpoint accounts for the largest morbidity welfare cost in all but one instance (Strukova *et al.*, 2006). Whilst the survey of studies is not comprehensive, this finding is notable since it holds even though it cannot be assumed that the modelling practices or the geographical contexts are similar. In practice, however, given the limited evidence-base, it is likely that the exposure-response functions used will be the same. For example, it is well-accepted that the epidemiological study of Abbey *et al.* (1995), which establishes a quantitative link between PM_{10}, SO_2 and NO_2 with new cases of chronic bronchitis, provides one of the most robust results available internationally, and is consequently widely adopted. In fact, the Strukova result is likely to have arisen from the different valuation procedure used in that study where, for morbidity, values for Disability Adjusted Life Years (DALYs) were used as a proxy for the pain & suffering component of WTP.

29. The paucity of policy applications in the water, waste and noise contexts do not presently allow a comparison of the importance of health endpoints to be made. In the case of water and waste, this may be the consequence of the difficulties that arise in modelling the pathways of pollutant emissions through to health impact; these pathways appear to be more spatially context-specific than in the modelling of air pollutant impact pathways. However, some recent comparative estimates have been made for EU25 countries (Scasny *et al.* (2009)). The authors find that total external costs of micro-pollutants, including heavy metals, formaldehyde, dioxins and a number of radio-nuclides, all as a result of health impacts, averaged about 1.5% of the external costs of the classical air pollutants over the EU27 countries, though it is worth noting that the evidence on monetary valuation relevant to these impacts (including *e.g.* IQ (Scasny et. al. (2008)) are rather speculative. Since the health impacts resulting from the classical air pollutants comprised around 90% of the total external costs in this analysis, it is unlikely that the health impacts of micro-pollutants total more than 2% of those health impacts associated with the (measured) air pollutants.

30. In the case of noise, policy analysis – where it utilises monetary valuation of the benefits – adopts values related to annoyance levels in the affected population, and it may be that specific health consequences of noise are subsumed into annoyance values. However, Defra, (2008), has undertaken a provisional analysis of the welfare costs of noise pollution associated with health impacts in the UK. It estimated that the combined welfare cost of heart disease, children's learning difficulties and tinnitus

associated with traffic noise is \$1.8bn annually. This compares with the annual welfare costs of health impacts in the UK associated with classical air pollutants – as estimated by Scasny *et al*. (2009) – of \$3.3bn. The health impacts of noise pollution are therefore equivalent to 55% of those of air pollution.

31. This partial comparison indicates that air pollution may currently be ranked as the most important cause of environmental health impacts, ahead of noise and micro-pollutants. However, the data does not currently exist to confirm this ranking over a wide range of countries, nor does it allow consideration of a wider range of health impacts, including those from climate change or water-borne bacterial disease. Thus, whilst an initial finding is that future research may well be targeted to valuation in the air pollution context, in different world regions, the importance of the gaps is currently unknown, suggesting that these gaps should also be addressed.

Table 13. Air Pollution-Related Health Impacts -- Physical Units & Costs -- EU15, in 1998

Impact	Unit	Physical Impacts	Unit values (€)	Damage Costs	Damage Costs
				[M. $€_{2000}$]	% of total
Years of Life Lost (YOLL)	[1000 years]	886	104760	88000	67.69
Congestive heart failure older 65	[1000 cases]	14.3	3260	48	0.04
Chronic bronchitis, adults	[1000 cases]	107	169330	18000	13.85
Restr. Activity days, adults	[1000 days]	110000	110	12000	9.23
Bronchodilator usage, adults	[1000 cases]	24600	40	1000	0.77
Cough, asthmatics, adults	[1000 days]	26100	4	1200	0.92
Lower resp. symptoms, adults	[1000 days]	9400	8	75	0.06
Bronchodilator usage, children	[1000 cases]	2950	40	120	0.09
Cough, asthmatics, children	[1000 days]	5100	45	230	0.18
Lower resp. symptoms, children	[1000 days]	4000	8	32	0.02
Chronic cough, children	[1000 epis.]	2280	240	550	0.42
Cerebrovascular hosp. Adm.	[1000 cases]	29	16730	470	0.36
Respiratory hosp. Admission	[1000 cases]	27	4320	110	0.08
Minor restr. Activity days, adults	[1000 days]	14000	45	640	0.49
Asthma attack, asthmatics	[1000 days]	390	75	29	0.02
Symptom days	[1000 days]	85000	45	3800	2.92
Total				**130000**	**100**

Source: Derived from Droste-Franke and Friedrich (2004).

32. The tables above indicate that the valuation of environmental health impacts exists, and – to date – has been undertaken in the policy context of air quality regulation, applied principally to the energy and transport sectors. However, it is also notable that policy applications are not abundant. Bureau and Glachant (2006), in a recent review of the role of monetary valuation of externalities in energy policy decision making, identify why this is the case. They find – through conducting a series of interviews with key government personnel in France, UK and USA – that whilst environment departments within central government produce guidance and recommendations on good practice in the use of non-market monetary valuation, (see *e.g.* USEPA, (1999), and USEPA, (2000)), such use is only mandatory in the US in conducting RIAs of prospective policies. Furthermore, Harrington and Morgenstern (2004), reporting on US practice, point out that where monetary valuation is included in a policy CBA, it is sometimes undertaken in the latter stages of the policy process, and often when key decisions have been made. The results of a CBA then act more to defend existing decisions.

Table 14. Selected Policy-relevant Studies -- Total Welfare-costs for Individual Air Pollution-related Health Endpoints
In original currency units and year

Study / Year/ valuation units + year	**Acute Bronchitis**	**Chronic Bronchitis**	**RHA/ per admission**	**CHA / Per admission**	**Asthma Symptom Day**	**Asthma Attacks**	**RAD / MRAD**	**Respiratory Symptom Days**	**ER Visit**	**Work Loss Day**
Abt Associates 2002. (M USD - 2000)	0.8	1740								
AEA Technology – 1998 (M Euro- 1998)		1200					447			
Chestnut *et al*, 2005 (M USD – 2000)	10 (Child)	4056	123	233					4	228
DEC NSW/ 2005 (000 AD - 2003)	1680 - 13400	52500 - 2440000	1480 - 3880	4380 - 11300		0 - 270	2870 – 4600			
Euston *et al.* – 2003 (M USD - 1999)		1149.38	22.23			274.23	204.14	222.77	16.58	
Office of Air Quality – 2000 (M USD - 1997)	<1	40					<1			<5
Stieb *et al.* 2002 (CD - 1999)			3770	4830	28		48	13	5610	
Strukova *et al.* – 2006 (B USD 2004)		0.026	0.01				0.076		0.026	
US EPA – 1995) (M USD – 1994)		974.0	11.3	9.4	56.9		147.0			
Zhou *et al.* 2005 (M USD – 2003)		116	1	6		1	81		2	

33. Bureau and Glachant (2006) find across the countries that the principal common obstacles to the use of non-market values, including of health impacts, in environmental policy decision making are the following:

- uncertainties surrounding the quantification of these values, (notably, but not only, the valuation of mortality risks);
- the cost of new valuation studies that may, or may not, reduce uncertainties associated with existing values;
- reservations about the ethics of such values, including the treatment of spatial transfer, and the distant future relevant to nuclear and climate change impacts;
- cost-effectiveness is the incumbent decision-making rule in many of the Ministries that would potentially use non-market values.

34. Part II addresses the first two of these bullet points, whilst Part III, in discussing value transfer practice, touches on the third point. The fourth point may perhaps be addressed over time as the responses of the research community to the first three points become more robust and are better understood. In relation to these points, a further suggestion[2] has been made that the underlying rationale for the use of economic efficiency criteria in public sector decision making, with its emphasis on the aggregate social welfare function, may be misplaced in the context of the decision-making within individual Ministries who can be argued as viewing their activities in the more narrow context of maximising welfare related to their own (sectoral) responsibilities, only.

Part II: The Current Empirical Evidence Base

Introduction

35. The primary purpose of this section is to provide an overview of the empirical literature on monetary valuation of environmental health impacts. Since a companion paper, OECD (2009), undertakes a detailed meta-analysis of values relating to mortality, the focus here is primarily on non-fatal health impacts, though some impacts considered here, such as cancers, may be fatal or non-fatal.

36. Prior to the review of the empirical literature, however, it may be helpful to provide a brief summary of the theoretical underpinnings of this literature. The following sub-section therefore outlines the theoretical basis.

Environmental morbidity valuation: Theory

37. The theoretical basis for economic measures related to morbidity effects of pollution is given by the health production and choice model and its variations (*e.g.* Cropper, 1981; Harrington and Portney, 1987; and Dickie and Gerking, 1991). We follow Freeman (2003) and present a simple model that examines the relationships among WTP for reduced pollution, reduced cost of illness, and defensive expenditures. In this model, health is measured by the number of sick days, (s), in any period of time, which *ignores the severity* of the illness and differences in the symptoms experienced. Among other determinants of the health status, the level of exposure to pollutants or the dose of the contaminant, (d), depends on the concentration of the pollutant, (c), and the amount of the averting activities, (a), undertaken to reduce the exposure to pollution. Additionally, the individuals can choose mitigating activities and treatments, (b), to reduce the health effects of a given level of exposure to pollutants. Examples of

[2] Sandy Hoffman, RFF, personal communication, 18 September, 2009.

mitigating activities, (b), include visiting a doctor or taking medicines to reduce a symptom, while examples of averting activities, (a), include staying indoors in days of high levels of pollution. The health production function of an individual can be formalised as follows:

$$s = s(c, a, b)\text{, with } \frac{\partial s}{\partial c} > 0 \text{ and } \frac{\partial s}{\partial b}, \frac{\partial s}{\partial a} < 0 \qquad (1)$$

38. Individuals maximise their utility function, (u), subject to their budget constraint. Utility depends on the consumption of a numeraire good, (X), normalised with a price of 1, leisure (f) and health. Formally:

$$\max \quad u = u(X, f, s)$$

$$\text{subject to} \quad I + p_w(T - f - s) = X + (p_a * a) + (p_b * b) \qquad (2)$$

where:

I non-labour income;

p_w wage rate;

T total time available;

p_a price of averting activities;

p_b price of mitigating activities;

and $\frac{\partial u}{\partial X}, \frac{\partial u}{\partial f} > 0$ and $\frac{\partial u}{\partial s} < 0$

39. First-order conditions for a maximum include: $\frac{\partial u}{\partial X} = \lambda$; $\frac{\partial u}{\partial f} = \lambda * p_w$; and

$$\lambda * \frac{pb}{\partial s/\partial b} = \frac{\partial u}{\partial s} - \lambda * p_w = \lambda * \frac{p_a}{\partial s/\partial a}$$, where (λ) is the marginal utility of income.

40. Pollution can affect utility through health, aesthetic amenities and odour, but in this simple model, pollution affects utility only through health. In this case, WTP for reduced pollution is the reduction in the cost of achieving the optimal level of health given a variation in pollution levels. The marginal willingness to pay (MWTP) for a reduction in pollution, (w_c), is given by the marginal cost of reducing the number of sick days associated with the reduced pollution. It is obtained by differentiating the indirect utility function $v(I, p_w, p_a, p_b, c)$ and solving for (w_c):

$$w_c = \frac{dI}{dc} = -\frac{\partial v/\partial c}{\partial v/\partial I} = -\frac{\partial v/\partial c}{\lambda} \qquad (3)$$

41. However, the effect of pollution concentration, (c), on utility consists of two components, the direct loss of utility associated with the illness and the opportunity cost of the time lost due to the illness, valued at the wage rate:

$$\frac{\partial v}{\partial c} = \left(\frac{\partial u}{\partial s}.\frac{\partial s}{\partial c}\right) - \left(\lambda * p_w * \frac{\partial s}{\partial c}\right) = \left[\frac{\partial u}{\partial s} - (\lambda * p_w)\right].\frac{\partial s}{\partial c} = \left[\lambda.\frac{p_b}{\partial s/\partial b}\right].\frac{\partial s}{\partial c} \quad (4)$$

42. Substituting equation (4) into equation (3) and using the implicit function rule we obtain the expressions for MWTP, or (w_c):

$$w_c = -p_a.\frac{\partial s/\partial c}{\partial s/\partial a} = p_a.\frac{\partial a}{\partial c} = p_b.\frac{\partial b}{\partial c}, \quad (5)$$

where $\frac{\partial s/\partial c}{\partial s/\partial a}$ and $\frac{\partial s/\partial c}{\partial s/\partial b}$ are the marginal rates of substitution between pollution and the other inputs in the production of health, which are equal at the margin in order to minimise the cost of producing health (or maximising utility). As can be seen in equation (5), MWTP can be calculated from the reductions in expenditures on either mitigating or averting behaviour measures taken to attain the original health status, *ceteris paribus*.

43. Freeman (2003) discusses the difficulties associated with the practical implementation of (5) and, alternatively, suggests another expression relating MWTP to observable costs of illness. The author suggests one initial step to obtain the demand functions for (a), $a*(I, p_w, p_a, p_b, c)$, and (b), $b*(I, p_w, p_a, p_b, c)$. These demand functions give the optimal quantities of (a) and (b) as functions of income, prices and pollution. By taking the total derivative of the health production function, one can estimate the effect of a change in pollution on illness:

$$w_c = \left(p_w.\frac{ds}{dc}\right) + \left(p_b.\frac{\partial b*}{\partial c}\right) + \left(p_a.\frac{\partial a*}{\partial c}\right) - \left(\frac{\partial u/\partial s}{\lambda}.\frac{ds}{dc}\right) \quad (6)$$

44. Equation (6) suggests that MWTP for reduced pollution is the sum of observable reductions in the economic value of reductions in sick-time and mitigating activities (the cost of illness), averting activities and the monetary equivalent of the disutility of illness. The marginal willingness to pay for an exogenous reduction in illness, (w_s), can be derived as a special case of (6) by supposing that averting behaviour, (a), is not possible, or is minimal, and that mitigation, (b), reduces sick days from its exogenous level, ($s*$), according to $s = f(s*, b) = s* - s(b)$. The analogue to equation (6) states that MWTP is the sum of the cost of illness (mitigation costs plus lost wages) and the monetary equivalent of the lost utility. Formally:

$$w_{s*} = p_w + \left(p_b.\frac{\partial b*}{\partial s}\right) - \left(\frac{\partial u/\partial s*}{\lambda}\right) \quad (7)$$

45. As can be seen in equation (7), the economic costs of the health impacts of air pollution can then be given by the sum of three different categories:

(i) *Resource costs*: represented by the direct medical and non-medical costs associated with treatment for the adverse health impact of air pollution plus avertive expenditures. That is, all the expenses the individual faces with visiting a doctor, ambulance, buying medicines and other treatments, plus any related non-medical cost, such as the cost of childcare and housekeeping due to the impossibility of the affected person in doing so;

(ii) *Opportunity costs*: associated with the indirect costs related to loss of productivity and/or leisure time due to the health impact;

(iii) *Disutility costs*: refer to the pain, suffering, discomfort and anxiety linked to the illness.

46. Two general approaches for valuing the benefits of reduced morbidity associated with environmental programmes are the cost of illness approach (COI) and the willingness-to-pay (WTP) approach. The first approach measures *direct* costs of morbidity, such as the values of goods and services used to treat the illness; plus *indirect* costs of morbidity, such as the value of forgone productivity. The loss of economic productivity is estimated based on the human capital theory, which uses discounted lifetime earnings as its measure of value, assigning valuations in direct proportion to income.

47. The cost of illness approach in general reflects the societal resource costs of illness and is often based on aggregated data. The WTP approach, instead, is based on individual data and assumes that the preferences of individuals can be characterised by substitutability between income and good health, that is, individuals make trade-offs between consumption of goods or services and factors that increase the consumer's health status. These trade-offs reveal the values individuals place on their health. In this paper it is assumed that the WTP measure accounts only for the disutility cost element (though it is recognised that some overlap may exist in empirical measurement if the components are not explicitly separated). The total social welfare cost is then estimated by summing the COI and WTP components associated with a given impact.

48. In practical valuation exercises there is of course some skill required to ensure that these component measures do not overlap and so result in double-counting. For example, the individual will include both financial and non-financial concerns in his/her assessment of loss of welfare unless the WTP measurement ensures otherwise.

49. For this reason, some studies – including those using stated preference methods – combine both components. A recent example is the study by Chestnut *et al.* (2006), in which the survey population had direct experience with the illness episode that caused an individual to be hospitalised, which could be a result of either a serious acute illness or an aggravation of a chronic illness. The authors included in the same questionnaire a series of COI questions that focused on the respondents' most recent hospitalisation, whilst the WTP questions referred to preventing or shortening a hypothetical future hospitalisation. The authors justified this procedure by arguing that it was necessary to make the WTP questions realistic and consistent with needs of policy analysis where *ex-ante* valuation is most appropriate.

50. Given the extra sampling effort necessary to obtain individual COI measures and the extra scenario-definition effort necessary to estimate WTP only for the disutility costs of morbidity effects, it seems that the WTP approach used to elicit the total economic costs associated with specific morbidity endpoints may be useful in avoiding the double-counting issues mentioned above. In this context, it is interesting to observe that the practice pursued most often in North American studies – as exemplified by the Chestnut et. al. (2006) study – of asking respondents for COI as well as disutility WTP, contrasts with most European practice that asks respondents only for the disutility WTP. One explanation for this

difference is the possibility that North American respondents often bear more of the costs of illness directly themselves and so, when giving survey answers, are more likely to consider these costs.

Environmental Health Valuation: Empirical evidence

51. Tables 1-6, above, illustrates the fact that there are a wide variety of health endpoints that may require monetary valuation if they are to be used in a decision-making framework such as CBA that requires a money metric. However, whilst the evidence presented above indicates that some value estimates exist and are used in decision-making contexts, it was also highlighted above that a number of uncertainties attendant with valuation of environmental health endpoints serve to constrain their use.

52. To illustrate this point, Rabl and Spadaro, (2005), assesses the extent of these uncertainties. He finds a wide spread of values, equivalent to a geometric standard deviation (σ_g) of 2 for mortality and chronic bronchitis, (CB), and 1.2 for hospitalisation. The distinction is based on the assumption that mortality and CB values are derived from non-market techniques, the data from which are derived, is more uncertain than that for hospitalisation whose valuation is based on market data. However, since all health endpoints are likely to have a disutility (pain and suffering) component which can only be valued using non-market valuation techniques, it may be argued that a σ_g of 2 should be assumed for all morbidity endpoints.

53. An example of the importance of this distinction in a policy application is the differential treatment of COI (market) and WTP (non-market) components of the health unit values utilised in the cost-benefit analyses of the ambition-levels assessed within the Clean Air for Europe (CAFE) Programme in 2001, which led to the proposal and adoption of the Thematic Strategy on Air Pollution in September 2005. Table 15 shows the effect of excluding, or including, the WTP component of the mortality and morbidity impacts in the benefits estimation. These are shown, relative to the costs of reaching the ambition level, of €71 billion. The results show that without any health WTP benefits, the B-C ratio is 0.6 Adding the morbidity WTP improves the ratio to 2, whilst adding the mortality WTP gives a ratio of 6. Thus, in this case, the WTP values help to determine the outcome of the CBA.

Table 15. CAFÉ CBA With/Without WTP Components of Health Impacts
€, 2005

Cost element	Unit	Total
Medical cost	€billion	0.38
Lost production cost	€billion	3.06
Crop losses	€billion	0.33
Materials	€billion	0.19
Total	€billion	3.96
Total	**as % of €7.1 billion cost**	**56%**
Adding in non-mortality WTP		
Non-mortality WTP	€billion	10.40
New total	€billion	14.36
New total	**as % of €7.1 billion cost**	**202%**
Adding in mortality WTP		
Mortality WTP	€billion	29.09
Grand total	€billion	43.45
Grand total	**as % of €7.1 billion cost**	**612%**

Source: Holland, Pye and Hunt (2006).

54. This section outlines a number of the methodological issues that underlie the levels of uncertainty that appear to exist in environmental health valuation. Whilst uncertainty in mortality valuation is clearly critical, its investigation elsewhere (OECD, 2009) allows us to focus on the derivation of other, non-fatal

health endpoints. In order to make the discussion manageable, we draw out the methodological issues using the prism of the valuation of a small number of morbidity endpoints. As was noted above, chronic bronchitis is regarded as the most important health impact resulting from the emissions of the classical air pollutants – particulates, sulphur dioxide, nitrogen dioxide and ozone – provisionally ranked the most important range of human health-damaging pollutants. The valuation of cancers is also given specific attention, given their perceived importance over a wide range of environmental media, including air, water and radio-activity, and the on-going concerns relating to emerging environmental hazards, such as electro-magnetic fields (EMF).

Cost of Illness

55. Whilst the medical and labour cost components necessary to estimate the COI associated with a given health endpoint are, in principle, measured by the use of market data, the measurement task is often not straightforward and – in the case of more serious illnesses – often ill-defined. An initial challenge is to ensure that the physical description of the health endpoint is sufficient for the length of treatment and the length of absence from work (if in the labour force) to be established. This information is generally not given by the definition provided by the epidemiological evidence used to quantify the impact in physical terms. As a consequence, more-or-less *ad hoc* assumptions are often introduced by the analyst (sometimes – but not always – in consultation with a medical practitioner) as the valuation exercise gets under way.

56. Similarly, the type of medical treatment required, how and where it is administered, and its effectiveness, all need to be agreed upon. It is notable that medical practices can vary dramatically between countries, and even within countries, for a given health condition. In addition, different discount rates may be applied to those treatment costs that occur over a number of years; this factor may be further complicated by changes in medical technologies, and their costs, over time. An example of the way in which these, and other, differences combine to produce a range of treatment cost estimates is given in Table 16, where a range of cost estimates for lung cancer treatment are presented.

Table 16. COI Studies on Lung Cancer: Medical Treatment Costs
€, 2005, PPP

Author	Country	Approach	Viewpoint	Discounting	Time-span	Lung cancer
Koopmanschap (1994)	Netherlands	Incidence Prevalence	Not specified	No	-	14,131
Evans *et al.* (1995)	Canada	Incidence	GOV	No	-	21,292
Berthelot *et al.* (2000)	Canada	Incidence	GOV	No	-	-
Wolstenholme Whynes (1999)	UK	Incidence	Hospital	Yes (6%)	4 years	-
Weissflog *et al.* (2001)	Germany	Prevalence	Sickness fund	No	-	33,744
Serup-Hansen *et al.* (2003)	Denmark	Incidence	?	Yes (3%)	-	15,231
Braud *et al.* (2003)	France	Incidence	Hospital	No	-	13,332
Chouaid *et al.* (2004)	France	Incidence	Healthcare payment	No	1.5 years	-
Vergnenegre *et al.* (2004)	France	Incidence	Healthcare payment	No	2 years	-
Dedes *et al.* (2004)	Switzerland	-	Health service expenses	?	2.5 years	-
Abal Arca *et al.* (2006)	Spain	Incidence	?	?	-	4,637
Scasny *et al.* (2008)	Czech Republic	Incidence	GOV	Yes (1%)	10 years	10,993

Source: Scasny *et al.* (2008).

57. In the case of Table 16, it is very difficult to undertake a convincing analysis of why the results of the studies differ because not all the relevant information is presented for all the studies. Nonetheless, it seems clear that many of the differences can be explained by the study method, *e.g.* whether the study was a clinical trial or adopted a population cohort, the type of lung cancer valued, (non-small cell or small cell), and the alternative assumptions made about the length of hospital stay; the total treatment period; the discount rate and the unit costs used. Since we do not know the type of lung cancer associated with each case attributable to the complex pollutants, and since it is not possible to easily discriminate between the studies on the basis of quality, a range encompassing the diversity of values identified – $€_{2005PPP}$ 4,600 to $€_{2005PPP}$ 27,800 per case – may need to be adopted in a monetisation exercise.

58. Opportunity costs associated with lost productivity resulting from a health condition are also more uncertain than might be expected. One problem, related to the definition of the endpoint, is to establish the extent to which the endpoint impacts those in the labour force. Again, an informed assumption as to the proportion of those impacted who are working may need to be made. The conventional approach – the human capital approach – measures the cost on the basis of the gross income lost during the time of absence from work (see *e.g.* Garattini *et al.* 2000). The value of productivity loss is usually calculated using per capita average wages. A problem with this is that unpaid work of people outside labour force (elderly, children, students, etc.) is omitted, though this problem may be mitigated by estimating a monetary value for unpaid work, such as housekeeping or volunteering, as was undertaken in a recent Danish study (Serup-Hansen, 2004). Other costs may, in principle, need to be accounted for, including the various costs associated with hiring and training of work force replacements and the indirect costs resulting from lower customer satisfaction and poorer quality of products or services. Again, the work by Scasny *et al.* (2008) on lung cancer illustrates the effect of these factors being included or omitted. They find a range of costs per case of $€_{2005}$ 27,000 to $€_{2005}$ 273,000.

59. On observing these differences in the practice of estimating costs of illness, the question arises as to whether and how commonly agreed assumptions need to be established. Common assumptions are clearly preferable for comparability of disease costs, particularly across countries. They are also important in establishing that best practice is adopted in future studies. It is also the case, however, that valuation studies are often undertaken to inform resource allocation decisions within specific national contexts where cost accounting rules relating to health treatment are already agreed and accepted within national decision-making frameworks.

60. To the extent that commonality in COI is required, some clear guidance on best practice can be offered. Most importantly, in a multi-country study the treatment of productivity losses as a result of a given disease needs to be related to an agreed definition of the disease, and its interpretation in the context of how it impacts on both paid and unpaid activity. The key aspect of the definition in this context is, then, the duration of this impact. Also important is clarification of who pays the productivity costs incurred and the issue of cost comparability across labour markets with differing structures and conditions. In the case of the latter, it seems reasonable for a multi-country study to take an average of wage rates across the countries, perhaps weighted by size of working population. This practice was adopted in the recent CAFE CBA in Europe,[3] and the air quality RIAs in the US. Alternatively, if national labour market data is not available, national GDP per capita estimates may be used. This latter measure has the merit of including, implicitly, non-market labour.

3 AEA Technology (2005) summarises the different unit damage costs that were then later used in the cost-benefit analysis in CAFE. Additional reference documents of the CAFÉ project can be found at http://ec.europa.eu/environment/archives/cafe/general/keydocs.htm.

Willingness-to-pay (WTP)

61. The sub-section above identifies a number of reasons why uncertainty is introduced into the measurement of the costs of illness. It is likely, as suggested by Rabl and Spadaro (2005), that the uncertainties attached to the WTP of disutility are still greater. Why may this be the case? The paragraphs below review the empirical evidence on the valuation of WTP in environmental health impacts, first through the use of revealed preference methods. Stated preference applications are then reviewed in more detail.

62. Before reviewing these sources of uncertainty, however, it is worth highlighting that whilst the uncertainties attached to COI and WTP are not trivial, the uncertainties attached to the quantitative estimation of other parts of the impact-pathway are of a similar dimension. Rabl and Spadaro (2005) investigate these relative uncertainties more formally; their findings are summarised in Table 17. They find – in the air quality context, using geometric standard deviation as the measure of uncertainty – that whilst the uncertainty associated with pollutant exposure modelling is a little less, that associated with the physical quantification of health effects using concentration response functions is comparable with that for morbidity or mortality valuation using contingent valuation methods. Indeed, it is probable that for environmental media other than air, where the physical modelling is relatively mature, the uncertainties attached to the physical modelling will outweigh those attached to the monetary valuation.

Table 17. Uncertainty of Health Damage Cost estimates per Kg of air pollutant emission using geometric standard deviation, (g)

Impact pathway component	**PM** σ_g	**SO_2 via Sulphates** σ_g	**NO_X via Nitrates** σ_g
Exposure calculation			
Dispersion	1.5	1.5	1.5
Chemical transformation	1	1.2	1.4
Background emissions	1	1.05	1.15
Concentration Response Functions			
Relative risk	1.3	1.3	1.3
Toxicity of PM components	1.5	2	2
YOLL, given relative risk	1.3	1.3	1.3
Monetary valuation			
Valuation of YOLL (VOLY)	2	2	2
Morbidity	2	2	2
Total	2.65	3.13	3.26

Source: Derided from Rabl and Spadaro (2005).

Revealed Preference

63. In cases where revealed preference valuation techniques are used, the econometric modelling and data demands are such that large parametric uncertainties are introduced, and are generally unresolved. Such studies include Agee and Crocker (2001), who – using a health production function valuation method – estimate smoking parents' substitution rates between their own consumption and own health, between own consumption and their children's home exposure to environmental tobacco smoke, (ETS), and between their own health and own children's health. Observed data on adult expenditures on health, smoking behaviours and child and adult ETS exposure response relationships are utilised. The study finds that the smoking parents sample values a ten percent improvement in their children's health roughly twice that of the same improvement in their own health and identifies a significant negative association between sample parents' assessed health of their child and that child's daily exposure to ETS. However, there is no comparison with the preferences of a parallel analysis of non-smokers that would have shown how preferences vary between such groups and inform applications of restrictions on public smoking.

64. As with a number of studies (*e.g.* Jenkins *et al.*, (2001); Mount *et al.* (2001) that use expenditure data on safety measures in transport to estimate marginal WTP to avoid injury, it is difficult to identify monetary values for the specific health endpoints derived from existing exposure-response functions. In addition, there remain the problems of avoiding joint products and reconciling disparities between subjective and objective risks associated with avertive behaviour.

Stated Preference

65. The majority of studies that have applied non-market valuation techniques to estimate environmental health WTP have taken the form of stated preference studies. Annex 1, which provides a reference list (and indicative results) of a range of recent studies that have valued non-fatal health impacts of air and water pollution, demonstrates that the vast majority of these studies have adopted the contingent valuation technique. However, whilst there appears to be a convergence in the type of non-market valuation techniques used in the health context, there remain a number of other methodological issues that have yet to be agreed upon and which serve as sources of uncertainty. This sub-section focuses on those issues that are health-specific. However, there are a number of issues generic to the practice of environmental non-market valuation – for example, the choice of payment vehicles and bid structures – where progress is being made via experiments in other policy contexts that will benefit future applications in the health context (see *e.g.* Bateman *et al.* (2004) for an introduction to these issues).

Characterisation of risk versus certainty

66. Methodological issues that are health-specific, or that are particularly relevant to the health valuation context, relate broadly to how the good (health endpoint) to be valued is defined. For example, the epidemiological quantification of pollution-related health impacts expresses these impacts as risks of occurrence. For any given health condition, the policy-relevant valuation measure is therefore the willingness-to-pay for a change, (usually a reduction), in the risk of occurrence. Ready and Navrud (2006) also argue that using certain outcomes "does not allow consideration of potential changes in defensive actions that the respondent might take, such as limiting activity during periods of poor air quality". However, whilst in principle we would wish the valuation exercise to be couched in terms of risk changes, the practice differs between studies.

67. In fact, there appears to be a split between practice in the contexts of valuing acute, mild, illnesses compared to those illnesses that are chronic and more severe. In the valuation of acute morbidity, where the illness lasts no longer than a few days and has well-defined beginnings and ends, and where the WTP is derived for the avoidance of the illness that would otherwise occur for certain, the practice is often towards the use of certain health outcomes in the survey design. This is the case in *e.g.* Ready *et al.* (2004a), where the authors value a number of acute respiratory illnesses associated with air pollution. 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. One justification for using (WTP to avoid) certain health outcomes is that the survey respondent does not have to understand risks, and changes in risks, which have long been found to be difficult concepts to communicate effectively (Corso *et al.* 2001). The task of the respondent is therefore made easier. A second justification is that using risk changes in the context of valuing a mild condition, such as a cold or sore throat, may result in negligible WTP values for a good that is not well-understood. Valuation under certainty therefore represents a more practically expedient, though less conceptually appropriate, approach to WTP elicitation for use in health impact assessments.

68. In the case of valuing chronic conditions, it is more normal for risk levels to be used. By way of illustration, Table 18 first summarises four contingent valuation studies that have estimated WTP to avoid lung cancer risks. It shows that the studies value risk reductions from a given baseline incidence rate. The

table also serves to show how the resulting values vary between studies, though whether this variation is a result of methodological differences (*e.g.* elicitation method, public or private good), geographical context, sample size, or other differences, is hard to identify. Inclusion of a fifth and final study, by Åkerman *et al.* (1991), further serves to illustrate the paucity of valuation studies that exist for a given health endpoint, additional to the range of techniques and methods adopted, that perhaps help to encourage the perception within potential user-groups that the uncertainty is currently too great for these results to be usefully adopted in policy appraisal.

Table 18. Studies that estimate the WTP to avoid lung cancer

Study ref.	Good valued	Location	Valuation method	Sample size	Results (mean €2005)
Jeanrenaud and Priez (1999)	95% risk reduction of contracting lung cancer	Switzerland	CV (Payment card) Private good	757	VSC 0.37m – 0.43m
Aimola (1998)	50 % risk reduction of death from cancer	Sicily	CV (OE & Payment card versions) Public good	89	VSL 0.44m
Jan *et al.* (2005)	50 % risk reduction of lung cancer	Taiwan	CV Private good	328	VSC 0.015 – 2.5m
Hammitt & Liu (2004)	2/100,000 and 8/100,000 risk reduction	Taiwan	CV Private good Acute = 2-3 years LE Latent = 20 years + LE	1200	VSL 1.75m (acute); 1.32m (latent)
Åkerman, Johnson and Bergman (1991)	50 % risk reduction of lung cancer	Sollentuna, Sweden	Avertive behaviour	317	VSL 0.26m (40-year old, 3% d.r.)

Definition of endpoint severity

69. A number of related methodological issues extend from this first one. A second issue that relates to the compatibility between the epidemiological quantification and its monetisation is the characterisation of the severity of the endpoint. For instance, where the endpoint defined by the epidemiology has varying degrees of severity – defined in terms of restrictions on activity or duration – it is preferable to define the endpoint in a similar way. This may be done, for example, by presenting the respondent with the range of severities, together with the probability of suffering each, or a time-profile of a typical case and its severity at different time-points.

70. In the case of chronic bronchitis, (CB), the exposure-response function based on the study of Abbey *et al.* (1995), identifies symptoms which are very light (persistent cough or phlegm during at least two months) compared to the severity levels implicit in the only monetary valuation studies undertaken in OECD countries (Viscusi, Magat & Huber 1991, and Krupnick & Cropper 1992). While some cases are mild and temporary, CB can be a truly debilitating permanent condition, making it impossible to work or lead a normal life. The monetary valuation of Viscusi *et al.* was based on severe cases, with a questionnaire that was applied to the general population. Krupnick & Cropper (1992) used a slightly modified version of the questionnaire of Viscusi *et al.*, but by contrast, they applied the questionnaire only to individuals who knew someone with CB. Assuming that their sample was representative, the results of Krupnick & Cropper thus implicitly assume the average distribution of severity levels. However, one difficulty in applying the paper of Krupnick & Cropper – and one that is applicable to a number of other morbidity valuation studies also – is that their primary purpose was the development of the valuation methodology rather than the provision of numbers that could be used for policy. In this case, the focus of the study was on testing the relative merits of using alternative trade-offs: risk-risk or risk-income in deriving WTP values.

71. The chronic bronchitis example above again highlights the paucity of studies for important morbidity impacts. It also raises a further methodological issue: that of knowledge of the health endpoint. The epidemiological, population-based, approach favours an approach in the valuation exercise that

randomises the application of the survey to respondents from across the whole population, irrespective of whether respondents have experienced the illness themselves or not. Randomising of the survey observations across the population also provides a justification for the utilisation of WTP values in policy appraisal since, ideally, they represent welfare-based preferences within a public policy decision-making framework. However, this contrasts with another theoretical basis of the valuation exercise which supposes that the respondent is fully informed of the nature of the good being offered and so is able to make rational decisions regarding their budget allocation. An assumption, therefore, is that if a respondent has experienced the illness, they are better informed as to what it entails in practice.

72. This assumption, adopted by Krupnick and Cropper, was also made in the study by Alberini *et al.* (1997). According to the authors, the advantage of using a respondent-defined episode is that the good being valued is meaningful to the respondent. In this study, in order to define the illness episode, respondents were asked to check on a list of symptoms – ranging from headache, runny nose to asthma – all symptoms they experienced in their last health episode. These symptoms characterised the nature of the episode (*e.g.* cold or not); the respondents' indication of how long each symptom lasted characterised the illness duration; and the restricted activities that respondents informed characterised the severity of the illness (*e.g.* work or school losses; days in bed). The WTP question emphasised that it referred to the same episode, and that the stated WTP would refer to all parts of the total economic costs of morbidity (pain, suffering, medical expenditures, time spent in doctors and recovery, missed work and leisure).

73. One potential problem of the respondent-led definition of the health condition is that it may diverge from the condition as defined by the epidemiology. Additionally, and as Alberini *et al.* point out, it may be difficult for a respondent to recall his/her most recent illness episode, if any. In this case, WTP values obtained from self-described illness episodes may be unreliable. Closely related to this, is the evidence, gleaned from a number of studies, that charts the relationship between the health status of the respondent and their WTP to avoid future episodes of the illness. In fact, this evidence is quite mixed; whilst Dickie *et al.* (1987) found no such relationship, a number of more recent studies, such as Johnson *et al.* (2000) and Ready *et al.* (2001), found that those who had recently suffered from the health condition in question offered higher WTP to avoid future episodes.

Scope sensitivity

74. A further methodological issue, closely related to the discussion of valuation of uncertain versus certain health outcomes, is that of the proportionality of WTP to varying sizes of risk change being valued. This issue has proved troublesome in the context of valuing mortality risk reductions, particularly where the risk changes are small relative to the baseline. In particular, many studies have derived WTP estimates that are proportionately insensitive to the size of the risk change (Hammitt, 2000). Whilst Hammitt shows that, from a theoretical perspective, proportionality in WTP over size of health risk changes is to be expected, a survey of the empirical literature (Hammitt and Graham, 1999) shows that in each of the fourteen studies reviewed which reported on how estimated WTP varied with the magnitude of risk change – six valuing non-fatal risks – none showed proportionality to the risk, whilst only five showed WTP to be significantly related to the size of risk change. More recent studies of morbidity valuation continue to meet this difficulty. For example, a study by Adamowicz *et al.* (2007) values non-fatal and fatal cases of microbial illness and bladder cancers resulting from risk reductions from improved quality of municipal drinking water in Canada, using CV and CE methods. Again, the WTP results fail the risk proportionality measure. However, the weight of empirical evidence asserts that there is no easy answer to this methodological difficulty; use of larger risk changes or certainty lessens the policy relevance of the values derived; small but policy relevant risk changes are not easily understood by the respondent and give rise to WTP estimates of dubious robustness.

Context

75. A further, apparently intractable, methodological issue is that of the appropriate treatment of context. Specifically, early US guidance – the NOAA[4] expert panel on contingent valuation – concluded that respondents could not reliably answer contingent valuation questions about environmental goods unless the hypothetical programme to provide the good is described in detail. However, it is argued that the risk involved in valuing health endpoints in the context in which the risks are generated is that the focus of respondents' preference formation can be deviated from the endpoints themselves to the cause of the endpoints. For example, Rozan (nd) found that the main reason that respondents gave for refusing to engage in an air quality programme that would prevent them suffering a number of health symptoms was that respondents did not think they were polluters and should not suffer the financial consequences.

76. Similarly, Navrud (2001) claims that respondents are distracted in their valuation of the disutility of different symptoms once air pollution is mentioned as one of the possible causes for the increase in the frequency of symptoms. On the other hand, it is recognised that focusing on the endpoints in a context-free approach may cause respondents not to take the hypothetical scenario and WTP questions seriously enough to provide reliable estimates for policy analysis. In addition, context-free WTP questions embed the risk that individuals' valuation are based on individual information and subjective references, while context-specific valuation is based on the same information, providing better control over individual responses (Rozan and Willinger, 1999).

77. A somewhat comforting result in this regard was that reported in Ready *et al.* (2004a), who found little evidence that the mention of the cause of the illness (air and water pollution) influenced respondents' stated WTP to avoid an illness episode. The authors conclusion is, however, undermined to some extent by the fact that they mentioned the cause of the illness but did not mention how the episode would be avoided – a potentially relevant part of the context. Additionally, Rozan and Willinger (1999) found that WTP for reducing symptoms caused by air pollution depends on the respondents being aware of the origin of the symptoms. The observed WTP of respondents who knew that air pollution was the origin of the bad health state was approximately 50% higher than the WTP of respondents unaware of the origin of the symptoms.

78. One reason for the context being thought to be potentially important is that, for public policy interventions, respondents may hold altruistic preferences, *i.e.* for those other than themselves. For example, the study by Adamowicz *et al.* (2007) speculates that the relatively high values found for the values of statistical life (VSL) are due to respondents including these preferences when formulating their WTP for the health benefits of improved quality of public water supply. Such altruistic motives are likely to be hard to isolate – never mind that the *type of altruism* is important in determining aggregation that avoids double-counting and so needs to be identified – suggesting that the dilemma between use of context and validity of WTP is far from being resolved. Nevertheless, it is also likely to be the case that people who suffer from a given illness are not necessarily concerned with the cause of their illness; in determining their WTP to avoid the illness the duration and severity of the condition itself may well be more critical.

Valuation of Health Endpoints relating to Children

79. Related to this discussion, the treatment of health endpoints suffered by children affects to recognise the impact of parental altruism. Since children do not have the necessary level of discernment to evaluate their preferences for reduced morbidity, nor the necessary knowledge to engage in economic decisions, the usual practice in valuing children's health is to observe parents' behaviour towards their children's health and safety. A summary of this body of empirical evidence is presented in Table 19. In the studies presented, the utility function is assumed to be for the household as a whole (unitary model), as

[4] U.S. National Oceanic and Atmospheric Administration.

opposed to a function for the individual (pluralistic model). The distinction between unitary and pluralistic models is thought to be a potentially important one in determining the WTP for children's health given the belief that intra-household resource allocation is often more complex than the unitary models allow. However, to date, this distinction has not been made on a sufficiently rigorous, or frequent, basis to be able to draw conclusions as to its real importance.

Table 19. Children's Health Valuation: Selected Stated Preference studies

Study Reference	Country of study	Nature of the risk	Study method
Viscusi *et al.* 1987	US	Insecticide and domestic hygiene-related	CV
Tolley & Fabian 1999	US		CV
Liu *et al.* (2000)	Taiwan	A cold	CV
Dickie & Ulery 2002	US	Air pollution-related acute illness	CE
Navrud 2001	Norway	Cough Day	CV
Dickie & Gerking 2001	US	Skin cancer	CV
Dickie & Brent 2002	US	Air pollution-related acute illness	CE
Dickie & Gerking 2003	US	Skin cancer	CV
Maguire *et al.* (2004)	US	Pesticide-related cancer	CV
Dickie &Hubbell (2004)	US	Asthma	CE
Dickie &Gerking 2005	US	Skin cancer	CV

Source: Adapted from Hunt and Ortiz (2006); CV=contingent valuation; CE=choice experiment.

80. One conclusion that has been drawn in recent reviews of this literature, across both morbidity and mortality (see *e.g.* Hunt and Ortiz (2006)), is that the WTP to avoid a given health impact to their children is greater than that to avoid the same impact on themselves. For example, a US CV study (Dickie and Messman, 2004), indicates 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. In this study, 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 (Navrud, 2008). However, as with Adamowicz *et al.* (2007), referred to above, the type of altruism (pure or paternalistic) drives consumers' behaviour (Chanel *et al.*, 2005) and determines whether these values should be used in cost-benefit analyses, or whether double-counting will occur. One potential, partial, solution may be to include in the survey questionnaire questions about parents' concerns about their children's health specifically, rather than their well-being more generally, thereby encouraging preferences to be determined according to paternalistic altruism that avoids double-counting.

81. There are other issues in the valuation of children's health that have yet to be explored in any depth in the existing valuation literature. These issues include: the uncertainty about the evolution of disease treatment over a child's lifetime and that therefore contributes to the uncertainty as to costs of illness; the discounting of chronic and/or latent effects; the definition of when to utilise childrens' or adults preferences, (for example in the case when the exposure is as a child but the manifestation is as an adult, and; treatment of productivity loss that may subsequently result from childhood illnesses.

Combining health endpoints within single survey instruments

82. In addition to the health-specific issues discussed above, there are those issues that are generic to many non-market valuation contexts but which are also manifest in the health context. Two such issues are: the number of "goods" – in this case health endpoints – being valued in a single survey instrument, and the ordering effects that might arise depending on the order in which these endpoints are presented to the survey respondent (see *e.g.* Bateman *et al.*, 2006).

83. Clearly, there is a tension between the interests of the study funder, who would ideally like new empirical data on more rather than less endpoints, and the increased costs of a longer survey instrument

that includes more endpoints. There is also a limit to the cognitive burden that an individual respondent may be expected to cope with before fatigue sets in. To an extent, the number that can be included will depend on the design of the questionnaire. However, in CV surveys, it is rare for more than five endpoints to be valued. And the problem of ordering is likely to become more important as the number increases.

84. By way of illustration of the ordering problem, and a possible solution, the study of Ortiz *et al.* (2008), is useful to reference. In this study, a contingent valuation survey was conducted in Sao Paulo, Brazil, to estimate the population's WTP to avoid one hospital admission (HA) and one emergency-room visit (ER) due to respiratory and cardiovascular diseases associated with air pollution. The authors report that a pilot survey identified that respondents tended to show higher WTP to the endpoint elicited first. A subsequent pilot study tested questionnaires that elicited one health endpoint only and it was found that the mean and median WTP estimates were not significantly different from the results in the pilot survey where both endpoints were elicited together. Given that eliciting only one health endpoint per questionnaire would reduce the sample size with no observed significant benefits, the final survey instrument returned to the format where WTP for both health endpoints were elicited in each questionnaire, but split the samples into different orders: first HA then ER; and; first ER then HA. Results were then generated combining observations in both formats (orders) to minimise the potential observed ordering effect.

85. A further generic issue manifest in the environmental health context is the part-whole problem, where WTP for component parts of a wider change do not sum to the WTP found for the wider change, itself. In the health context, this issue may potentially arise if there is a discrepancy between the summed WTP for acute health impacts that are in actual fact the manifestation of a chronic health condition. The question for the researcher is then whether to value the acute symptoms alone, the chronic condition alone, or whether to combine both. One possibility, currently being investigated in the EC DG Research-funded HEIMTSA project, is to combine these effects in the valuation of chronic obstructive pulmonary disease, (COPD), in a "chained" approach similar to that used in mortality valuation by Bateman *et al.* (2009). For example, an acute symptom, such as a cough day, is valued initially before a trade-off is made against the COPD endpoint where the respondent is asked to find a point where he/she is indifferent between a) a certain cough day, and b) a given % chance of full health and a given % chance of COPD. The respondent would be able to adjust the two % chances (adding to 100%) to the point of indifference. The WTP for COPD could then be derived by comparing the weightings of the cough day to COPD with the WTP to avoid the cough day.

86. This method is similar to the risk-risk trade-offs adopted in other morbidity valuation studies, including Krupnick & Cropper, (1992), mentioned above, who find a greater stability in such trade-offs than in risk-money trade-offs. It is suggested that respondents find that the risk-risk trade-offs are easier to make than risk-money trade-offs between goods that they find too dissimilar.

87. A different point of departure for morbidity valuation is taken by Bachmann (2006) who develops an approach to jointly value morbidity and mortality risks. Given disease events are converted into Years Of Life Lost (YOLL) equivalents by using so-called disability weights.[5] The disability weights are derived from person trade-off (PTO) surveys administered to health care providers as part of the Global Burden of Disease study by the WHO (Murray and Lopez 1996; Murray and Acharya 1997).[6] The YOLL-

5 DALYs were adjusted so as to not employ discounting and age weighting (*i.e.*, the loss of a young or old person counts less than a loss at midlife). Thus, Bachmann (2006) did not value DALYs as reported by the WHO.

6 For some health endpoints, no disability weights were available. Hence, morbidity risks were assigned to two categories according to a recommendation from an International Life Science Institute (ILSI) panel reported in Pennington *et al*. (2002): Probably irreversible / Life-shortening effects (category 2) or Reversible / Non life-shortening effects (category 3). Classified health endpoints were assigned YOLL-equivalents that were 1/10th or 1/100th of an average cancer (belonging to category 1: Irreversible / Life-

equivalents in turn are then valued with a Value Of a Life Year (VOLY) - a metric used for monetary valuation of mortality risks. Based on VOLYs of €40 000, example values are given in Table 20.

Table 20. Exemplary YOLL-equivalents and monetary values

Health endpoint	YOLL-equivalents per case	Monetary value per case
Skin lesions	1.28	€51,200
Kidney damage	1.28	€51,200
Affected enzyme (lactate dehydrogenase) function	0.128	€5,120

Source: Table 7-7 and p. 199 in Bachmann (2006).

88. There are two main disadvantages of this approach.[7] First, the person trade-off survey to convert morbidity phases into YOLL-equivalents was conducted with health care providers. This contradicts the principle of welfarism, since the preferences of health care providers may not correspond well to the preferences of the general public. Second, when deriving a monetary value for mortality risks a dread factor may play a distinctive role that is not applicable to preferences related to morbidity risks. However, a major advantage of the approach is that one could quantify the costs of most morbidity endpoints in a rather straight-forward manner if there were a generally accepted monetary VOLY for a YOLL (-equivalent).

Part III: Practical Use of the Evidence Base

89. In this section, we review a series of issues that relate to the use of the unit values derived in primary valuation studies in practical policy applications. The most obvious are those relating to spatial and temporal transfer. However, Ready and Navrud (2006), in their recent review of morbidity value transfer, highlight that a number of the methodological issues identified in the previous section also constitute transfer issues. For example, the discussion about the possible importance of context in determining WTP can be re-formulated in terms of being a question about the legitimacy of transferring a value from the original study context to another, policy, context. The extent to which such a transfer is reasonable may then be decided by the balance of evidence relating to whether context is a significant determinant of morbidity WTP values. Similarly, questions of transfer legitimacy apply to whether WTP differs between those who are more or less familiar with the health condition of concern, discussed above, or between those who are younger or older.

90. Ready and Navrud (2006) also review the evidence on whether it is legitimate to transfer the value of one episode, or duration of illness, to an illness consisting of a number of episodes or time-periods via a simple multiplicative procedure, or whether there is a more complex pattern. They point out that the theory is not clear whether WTP to avoid additional symptom days should increase at a less than or greater than proportional rate as the number of additional days to be avoided increases. The latter may be the case if health, measured as the number of days in a year the individual does not experience symptoms, is a normal good with decreasing marginal utility. However, if health is viewed as something that an individual

shortening effects), respectively. YOLL-equivalents (including mortality outcomes) for average cancer are estimated at 12.8 years.

7 A third disadvantage concerns the valuation of mortality risk. In the DALY concept, the highest life expectancy at birth observed on earth is assumed to apply to all countries. This occurs in Japan. Even though the life expectancy at birth will be somewhat lower in several OECD countries, the life expectancy of a person affected by a disease is conditional on his or her actual age. This conditional life expectancy at a certain age might in fact come closer or even exceed the life expectancy at birth in Japan. In principle, the presented approach could be adapted to allow for country-specific life expectancies at a given age (and for a given gender etc.).

either has or does not have, then the marginal disutility of additional ill health may be low, once health status drops below some threshold.

91. Empirical evidence is also mixed. Dickie and Gerking (2002), find that respondents tend to state WTP to avoid acute illness less than proportionately with regard to the duration of the endpoint and with the number of symptoms avoided. That is, WTP per episode avoided is lower when respondents value avoidance of several episodes of an illness together, instead of just one (Ready *et al.*, 2004a). This means that if unit values computed for one symptom-day are applied to each symptom-day of multiple-day, or multiple-symptom, episodes, benefits would be overstated. The estimate of benefits is also therefore sensitive to whether a few people experience many days of a symptom or a large number of people experience one or a few symptoms-days (Alberini *et al.*, 1997). However, exposure-response functions linking air pollution to symptoms cannot precisely estimate the distribution of the person-day illness reduction among the population. Thus, we cannot know whether the predicted health impact relates to five episodes experienced by one individual or one episode experienced by five individuals (Ready *et al.*, 2004a). The picture is also complicated by the fact that the valuation evidence is equivocal. For example, Tolley *et al.* (1994) found that WTP to avoid 20 days of severe angina was over three times as large as WTP to avoid 10 days, implying increasing marginal value of duration (or, equivalently, decreasing marginal value of health). Ready and Navrud (2006) argue that the design of stated preference surveys may not currently allow resolution of this issue. Certainly, it is hard to reach any conclusion without a substantially larger evidence-base on which to draw.

Value Transfer

92. More generic questions relating to the legitimacy of transferring values from one context to another are discussed in the broader benefit transfer literature. Thus, in the following paragraphs recent developments in this literature are summarised, with specific reference to the environmental health valuation literature.

93. Benefits Transfer (BT), or alternatively Environmental Value Transfer, (values being monetised non-market costs or benefits associated with environmental change) is 'The use of existing information designed for one specific context to address policy questions in another context' (Desvousges *et al*., 1998). Or 'the transposition of monetary environmental values estimated at one site (study site) through market-based, or non-market based economic valuation techniques to another site (policy site)' (Brouwer, 2000).

94. The two main approaches to BT (Navrud, 2002) are:

(1) Unit value transfer
- Simple unit transfer
- Unit transfer with income adjustments

(2) Function Transfer
- Benefit function transfer
- Meta analysis

95. Simple unit transfer is the most straightforward approach to adopt and assumes the impact experienced by an average individual at the study site, and his/her welfare loss, is the same as that which will be experienced by the average individual at the study site. Such an approach is perhaps least appropriate regarding transfers between countries, hence the application of adjustments for income.

96. Unit value transfers of both varieties assume equivalence in preferences, environmental, cultural and institutional conditions between study and policy sites, whilst in reality these conditions will rarely hold. Function transfer attempts to account for this by transferring the entire benefit function that will

include much more information. Meta analysis takes this a step further by aggregating and analysing the results from several valuation studies together using multiple regression techniques. Meta analysis can help in the understanding the significance of specific factors in explaining differences in valuation outcomes across studies. According to Stapler and Johnston (2009) 'One of the potential advantages of MRM {the authors refer to meta analyses in their most usual form as meta-regression models MRM} benefit transfer relates to its capacity to generate benefit functions that are more broadly applicable and less sensitive to the attributes of individual studies, leading to the potential for reduced transfer errors' (p. 228).

97. The use of benefit transfer derives from the fact that whilst most researchers would advocate the use of a primary study in order to achieve an accurate valuation, this approach tends to be high cost. However, there is continuing debate in the field about the most appropriate methodologies for achieving reliable and valid value transfers. Reliability (of a transfer estimate or function) may be regarded as a necessary, but not sufficient, condition for validity. Reliability is affected by the usual criteria for the achievement of a sound empirical study, that is, sample adequacy, sound economic methods and statistical techniques, and the reliability of the data used to supplement estimated value functions. It may therefore be regarded as unwise to assume that unadjusted value transfers would have a high degree of reliability. Pearce (2000) notes that 'As a general rule, there is little evidence that the conditions for accepting unadjusted value transfer hold in practice'. He suggests incorporating an adjustment for the income elasticity of WTP. Others (for example Brouwer and Spaninks, 1999) suggest that adjusting transfer values for purchasing power parity (PPP) would be more appropriate in the international context, and this appears to now be the accepted form for transfer values within benefit transfer functions and for values within meta-analyses. However, Ready *et al.* (2004b) suggested that PPP may not be sufficiently sensitive in some contexts. In their 5-country study of morbidity valuation which tested the reliability of transfer values between countries, they made a further adjustment to account for the higher cost of living in major cities.

98. The validity or accuracy of transfers has traditionally been studied using convergent validity methods by conducting surveys at a number of different sites, thus enabling researchers to treat all sites as both 'survey' sites and 'policy' sites (see for example, Brouwer and Bateman, 2005; Rozan, 2004; Brouwer and Spaninks, 1999; Ready *et al.*, 2004b; Pearce, 2000; Downing and Ozuna, 1996; Kirchoff *et al.*, 1997). The results have not been totally conclusive on the issue of (a) whether transfers broadly are valid, or (b) whether transfer of unit values or transfer of benefit function has greater validity, but there seems to be broad convergence round the view that transfer of benefit function produces lower error rates than transfer of unit values, unless the study and policy sites are virtually identical (Rosenberger and Loomis, 2003). This clearly has particular implications for inter-country transfers. Additionally, transfer of a benefit function potentially has more explanatory power, which, for many policy purposes, is important. Brouwer (2000), in his review of a number of relevant studies, pointed out that whilst benefit function transfers potentially have more explanatory power than unit value transfers, the (then) current state-of-the-art was such that the explanatory power of the models used to specify differences in WTP was usually low, accounting on average for only about 30-40% of variance in stated WTP.

99. Firstly, and most salient to the concept of the construct validity of a benefit transfer function, is that of the estimated monetary values themselves. As Brouwer (*op cit*) notes "How can environmental values be reliably predicted across sites and people if currently much, if not most, of the variability of the values in the original studies cannot be explained?" (p143). Part II, above, highlights a number of reasons why such variability continues to exist. That notwithstanding, Brouwer (*op cit*) attempts to define 'a protocol for good practice' that outlines a number of steps to take in order to give the required attention to context and motivation, thereby improving not only reliability, but also validity:

Step 1 – defining environmental goods and services
Step 2 – identifying stakeholders
Step 3 – identifying values held by different stakeholder groups

Step 4 – stakeholder involvement in determining the validity of monetary environmental valuation
Step 5 – study selection
Step 6 – accounting for methodological value elicitation effects
Step 7 – stakeholder involvement in value aggregation

100. Bateman *et al.* (2006) note that it is also important to define the extent of the market for a public good: There is political jurisdiction, that is, the geographical/administrative area or the economic jurisdiction, that is, those holding economic values regarding the project. Understanding of this is necessary for the calculation of aggregate WTP estimates. Ready and Navrud (2006) make the same point that it is critical to determine the geographical extent of the population holding values for the environmental good. Brouwer and Spaninks (1999) also allude to this issue (defining the relevant population) in their concept of 'distance-decay'. They tested its relevance and found that it can confound results if not adequately controlled for.

101. Pearce (2000) too notes that "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 should be supplemented by social surveys at the policy site." (p10). Pearce (*op cit*) identifies a number of the contextual variables that should be included as potential explanatory variables in a benefit transfer function specifically for valuing life and health risks. These include the precise nature and quantity of the good (exact specification thereof, *e.g.* a cough of one day's duration), characteristics of the population such as age, gender, health status, culture, and attitudinal variables. Van Houten *et al.* (2006) provided a demonstration of this using a meta-regression analysis. Their findings showed that WTP estimates for avoided acute health effects varied systematically with respect to key explanatory variables such as age, average income, and various aspects of health status.

102. In general, recent studies and meta-analyses demonstrate a more consistent attention to a number of the socio-cultural and attitudinal variables identified above. Stapler and Johnston (2009), for example, explicitly investigated a wide range of 'methodological covariates' using a meta-regression model on a large dataset. Although their results did not highlight highly significant differences in transfer error, they noted that their approach was salient in understanding the patterns of impact of the methodological variables, and thereby, the sources of transfer error.

103. A different approach to examining the validity of benefit transfer was taken by Kristofersson and Navrud (2005). They argued that the classical hypothesis used in investigating the validity of benefit transfer has major drawbacks. Researchers generally take as the null hypothesis that there will be no difference between a primary study result and a benefit transfer estimate. Kristofersson and Navrud suggested that, given that the underlying assumptions of this approach are that there is equality of the environmental goods and that this is often unlikely to be the case in reality, then the null hypothesis of no difference is not appropriate. They therefore proposed the concept of equivalence testing, which tests the null hypothesis of difference, and can also include the concepts of statistical significance and policy significance. They suggest that this is more in line with the theory of environmental valuation.

104. Finally, Loomis and Rosenberger (2006) summarised three general criteria that are necessary for valid and reliable benefit transfers. These are commodity definition comparability, market area comparability and welfare measure comparability. This last refers to the necessity of matching the welfare measure required for the policy analysis with that available in the literature. Loomis and Rosenberger also suggested that researchers should be encouraged to develop the reporting around these criteria so that valuation databases contain more comprehensive accounts of relevant contextual information. Clearly such a development would facilitate future benefits transfer work.

International benefit transfer

105. A number of researchers have explicitly studied international benefit transfer (for example, Ready *et al.*, 2004b; Pearce, 2000; Rozan, 2004; Ready *et al.*, 2006). Rozan (2004) used the approach of transferring the benefit function in a study comparing WTP for improved air quality (including the associated health benefits) in France and Germany. This study was effectively an intra-site transfer as the valued environmental good (air quality) was exactly the same in both the study sites. Results showed that in a test of the validity of the benefit transfer, the hypothesis of equality between transferred and predicted WTP was rejected. Furthermore there was an error rate of around 30%. Rozan argued that such an error rate might be acceptable for some policy uses (for example, cost-benefit analyses), but not for others (for example, the design of compensation schemes).

106. Pearce (2000) reports error rates for benefit function transfer of + or – 40% and implies that for a cost-benefit policy context this may be acceptable. Ready *et al.* (*2004b*) also report 'that the majority of transfers result in errors less than 50% (p80), and make the same point about acceptability, that is that the extent to which this error rate is acceptable depends on the policy context. Brouwer (2000) and Brouwer and Spaninks (1999) both review a number of studies whose aim was to test the validity of benefit transfer. Their analyses resulted in a 27% error for unit value transfers and 22% for benefit function transfer. Ready *et al.* (2004b) found little difference between unit value transfers, adjusted unit value transfers (adjusted for income differences) and benefit function transfer, error ranges for all three falling in the range 20-40%. Brouwer and Bateman (2005), like Brouwer and Spaninks, found that the transfer of benefit functions between dissimilar contexts produced a lower error rate than the transfer of mean values.

107. Given the foregoing discussion about the need for PPP adjustments to WTP, and the need to understand attitudinal, socio-economic and cultural context, a further problem with international benefit transfer is highlighted. When attempting to transfer benefit values from developed to developing countries, there is often a lack of reliable data at the policy site (for example, population census data) to input to a function analysis. Clearly this will impact on the ability to build a comprehensive benefit function model, and the consequent uncertainty ranges and error rates for such transfers.

108. On benefit transfer, we can conclude that there seems to be some convergence of opinion that transfer of benefit function might be the more appropriate methodological approach to benefit transfer unless all variables have a very high degree of equivalence. Meta-regression techniques can also contribute considerably to an understanding error variance. It must also follow therefore, that transfer of a benefit function is likely to produce better, or more valid results in the international context. Building models that account for a number of contextual factors, including a range of demographic, cultural and attitudinal variables, can improve benefit transfer functions, and this applies critically when using BT in policy relating to health. Furthermore, the appropriate adjustments, such as conversion to PPP must be made to WTP valuations.

109. Many researchers active in this field have made pleas for others to adopt a common, standardised approach to the design and reporting of primary studies in order to facilitate more accurate benefit transfer. A number of areas of investigation (contextual variables, as above) have been outlined, and this approach to modelling is still being refined. It has also been noted (*e.g.* Bateman *et al.*, 2006) that it is vital to collect data on non-respondents, including their spatial distribution, in stated preference surveys, in order to account for self-selection and/or protest biases.

110. Application of benefit transfer methodology to the environmental-related health arena is still under-developed. Much of the work in refining the methodology comes from the environmental economics literature, where WTP (or WTA) decisions focussed around amenity loss or disutility are much less emotionally loaded than WTP decisions related to health. It is therefore unsurprising that the development

of benefit transfer function models, which are inherently likely to be more complex, is still evolving. In addition to the repeated imprecations in the literature for a standardised and comprehensive protocol for the conduct of primary research, there is also (of course) an identified need for more high quality primary research.

111. In order to illustrate what morbidity valuation stated preference studies actually comprise of, the box below provides further detail on one study – the Ready *et al.* 2004a,b study – that has recently explored a number of the methodological issues outlined in the sections above. The box, derived from Navrud, (2008), serves to show how health conditions are characterised in valuation studies; it also indicates how a number of methodological problems are tested in practice.

Box 1. Morbidity valuation case study: The Ready *et al.* 2004a,b study

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]

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, and 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).

The study 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.

Mean WTP values for each episode for each country, converted to UK £, 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.[9]

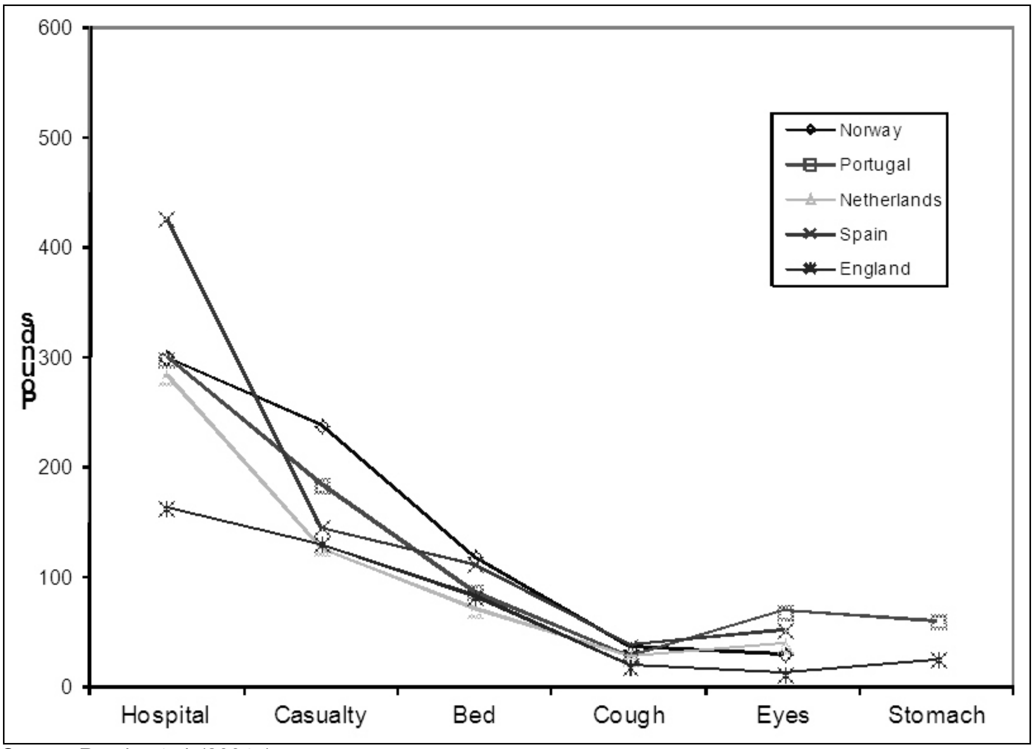
These results are somewhat counterintuitive, given differences in income among the countries. Spain and Portugal have much lower mean real incomes than the three Northern European countries, yet these two countries generally have intermediate to high WTP values relative to the other countries. However, 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

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 This study was later reproduced in both France and the Czech Republic (by Brigitte Desaiges *et al.* and Scascny *et al.*, respectively), which yielded even lower WTP to avoid these illness episodes (and lowest for the Czech Republic).

experience with symptoms included in the episode descriptions. 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. On this basis, the pattern of results is clearer. 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).



Source: Ready *et al.* (2004a).

Table 21. Ill-Health Episode Descriptions used in Ready et al (2004a).

Episode Name	Epidemiological End Point	Description
EYES (E)	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 (Co)	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 (S)	1 Work-Loss Day	One Day of persistent nausea and headache, with occasional vomiting. Some stomach pain and cramp. Diarrhea at least twice during the day. Patient is unable to go to work or leave the home, but domestic chores are possible.
BED (B)	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 (Ca)	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,	Admission to a hospital for treatment of respiratory distress. Symptoms

(H)	COPD, pneumonia, respiratory disease and asthma	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: All episodes are endpoints in exposure response functions for air quality, with the exception of "Stomach", which is linked to water pollution. COPD = Chronic Obstructive Pulmonary Disease.

Part IV: Concluding Comments

112. Monetary valuation of the health effects resulting from environmental hazards may be seen to offer an alternative way in which to communicate the extent of these effects, either to specific stakeholder groups or to the wider public. The power of the message that is being communicated may be increased by linking the health effects to specific causes and to consideration of alternative forms of mitigation, perhaps through policy intervention. In the case of the latter, estimates of the health benefits resulting from mitigating policies may be compared to the costs of these policies and so add a further dimension to the policy-makers' decision-making tool-kit.

113. However, these advantages will, in all likelihood, only be realised if the practice of health valuation is regarded as being both sufficiently robust and legitimate. A number of issues – principally methodological – have been outlined in Parts II and III of this paper that reflect the state-of-the-art regarding robustness. Whilst it is not necessary to repeat this discussion here, it may be useful to summarise the lessons that have been learnt so as to inform policy-makers in their commissioning of new valuation studies or their use of existing values.

1. Coverage of all welfare components

114. Any environmental health valuation application should look to include both the social costs of illness – including costs of avertive behaviour, treatment costs and lost productive time – as well as the willingness to pay to avoid (or willingness to accept compensation for) pain and suffering associated with a given health effect. There are advantages relating to consistency of undertaking a common survey for both the COI and WTP components. However, many OECD countries have health data systems that allow treatment costs to be constructed for individual health conditions. Prior to this, there is a need – particularly in multi-country studies – to establish a working definition of the specific endpoint, according to its severity, duration and its mode of treatment.

2. Revealed vs. Stated Preference

115. Revealed preference techniques have the advantage of being based on observed behaviour. In policy contexts, where it is not thought necessary to value specific health endpoints, and where there is data – and econometric expertise – available, these techniques may be appropriate. However, much environmental health impact assessment is undertaken through modelling individual sources and pollutants, and individual health endpoints, thereby allowing the numbers of cases to be estimated. In this instance, stated preference techniques are more easily tailored to valuation of individual health endpoints. It seems sensible to build on the (small but growing) body of knowledge that already exists in applications of these techniques within the environmental health context.

116. Whilst most applications of the stated preference method in this context have utilised the contingent valuation format, there is some experience with choice modelling (see *e.g.* Johnson *et al.*, 2000; Dickie and Ulery, 2002) which reflects a wider trend towards choice modelling in a variety of other environmental contexts. An advantage in the health context – as suggested by Johnson *et al.* – is that where the attributes utilised are generic in describing health conditions, WTP estimates may be able to be

transferred, adjusted for the level of the attribute appropriate to another health condition. In other words, it allows for the construction of a health-attribute benefit transfer function. However, as argued by Hanley *et al.* (2003), the empirical case for this is, as yet, unproven.

3. Alignment with epidemiology

117. In the case of many health endpoints, there may exist trade-offs between consistency in the definition of the endpoint available from the exposure-response functions used to quantify the impact in physical terms and the definition likely to be most clearly understood by a survey respondent. Examples of this include the definition of health outcomes in terms of certainty rather than risks, the definition of their severity levels, and the use of respondent-defined rather than researcher-defined endpoints. As the discussion above intimates, these trade-offs are not easily resolved. However, at the least, the policy-maker should be aware of what is at stake in making the trade-offs; uncertainty in the physical quantification versus uncertainty in the valuation quantification.

4. Scope sensitivity

118. The problem of respondents' (in)sensitivity, in their WTP valuation, to scope to the small risk changes likely to be relevant to most policy interventions is common to both mortality and morbidity impacts, and has been subject to a variety of different communication methods. As with (3) above, it appears that there are trade-offs that have to be made between the reliability of the WTP estimates and their fitness for purpose. It is to be hoped, additionally, that technology developments will allow more successful means of communicating risk changes to be developed through greater software ingenuity.

5. Context

119. The question of whether – and how – to represent the policy-making context in a survey design may to an extent be case-specific since there are often preferences within funding organisations for a certain treatment of context independent of the findings of the recent literature. The literature at present is in any case equivocal on this subject; one useful piece of generic analysis would therefore be to undertake an (ideally quantitative) assessment of the effect that context has on the composition and reliability of a range of existing WTP estimates.

6. Valuation of Children's' health endpoints

120. The range of divergent approaches explored and utilised in the EC DG Research VERHI project, and the range of WTP estimates subsequently generated for mortality impacts, reflect the fact that there is little consensus yet in methods to value children's health endpoints. This may be seen to be a consequence of the difficulties in this area of valuation that simply exacerbate those existing in health valuation generally, together with the fact that so few applications have been made to date, particularly for morbidity endpoints. A great benefit, however, of the VERHI project is that it was able to pool regionally disparate expertise. It is therefore to be hoped that such a critical mass of know-how can be exploited in subsequent studies in the near future, rather than being dissipated again by time and geography.

7. Ordering

121. There is a potential trade-off between the number of WTP estimates that the policy would like data on, and the quality of the estimates (that may be lowered as a result of too high cognitive burden on survey respondents) or the cost of larger sub-samples. Prior to commissioning, the policy-maker therefore needs to ascertain: a) which health endpoints are of relevance to his decision; b) whether existing valuations of these endpoints are good enough to transfer reliably to the policy context, and – if not; c) what are the least number of endpoints for which new WTP values are needed? If, subsequently, it is decided that more than one endpoint needs to be valued, the existing research findings suggest that the WTP for what is likely to be the most important endpoint should be valued first. If it is not known which endpoint is likely to be the most important, sub-samples (of sufficient size) should be created to enable the order to be varied.

8. No of episodes

122. Neither the theoretical or empirical evidence gives clear guidance as to whether it is reasonable to multiply the WTP derived for suffering from a health condition for one time period by the number of time periods over which the condition is likely to last. Survey design should therefore be led, in the first instance, by what the epidemiological literature (or personnel) reveals. However, there is likely to be little alternative other than undertaking rigorous pre-testing of this issue for the endpoints of interest.

9. Value transfer

123. In the case of value transfer, the initial trade-off – as indicated above – is between the "value-added" of a new empirical exercise (including definition of a tolerable error margin threshold) and the potential cost entailed. If, for a given endpoint, it is determined that the cost is likely to exceed the value-added, then value transfer procedures are likely to be adopted (though clearly, the extent of these will determine the value-added of the potential primary valuation study). Once it has been decided to rely on existing WTP estimates, the secondary trade-off to be made is the extent – or sophistication – of the transfer procedure. It was outlined above that one might expect greater specification of the benefit function, constructed from the data of one or more studies, to reduce the error introduced by the transfer. However, the evidence in general, and in the environmental health context in particular, is equivocal, though – it should be said – limited. In particular, the Ready *et al.* (2004b) study of morbidity endpoint valuation in five European countries finds no relationship between specification and error margin. In expanding the evidence base, the need for similar multi-country studies is perhaps most acute. In the absence of these, it is suggested that function specification should be limited by budget alone and results presented and compared with those using less sophisticated transfer procedures.

124. Our review suggests that the limited volume of evidence on which to base best practice is at least as limiting as the apparent methodological challenges in defining and targeting the sources of uncertainty in environmental health valuation. Funding agencies therefore need to appreciate that there is a (further) trade-off to be made in resource allocation between meeting immediate policy needs, and exploring outstanding methodological issues in greater depth. On the other hand, the research community has an obligation to focus its efforts on research questions that are likely to inform policy needs in either the short or long term.

125. Part I of this paper suggests that obvious examples of where there is a short-term policy need are in areas of environmental health hazards that appear to be relatively serious in aggregate. These areas include heart disease associated with noise and COPD and chronic bronchitis associated with the classical air pollutants. In parallel with these research foci, we would argue that the issues of altruism and risk communication should be principal areas of research with potentially wider application in the longer term.

126. At the same time, commissioning policy-makers have to be responsive to emerging environmental health issues. Prominent amongst these are the health impacts of climate change and nano-particles. In the case of climate change, valuation of health impacts may in principle inform both mitigation policy, including the assessment of ancillary benefits from reduced air pollution, and adaptation strategy. The valuation of health impacts of climate change brings new challenges: in addition to tackling a number of endpoints that have been given little attention – such as injuries and mental health impacts of flooding, and vector-borne diseases, including malaria – potential methodological difficulties include consideration of evolving preferences and WTP under long-term socio-economic scenarios. Ethical issues that arise in treating health impacts of climate change include the determination of long-term discount rates to be utilised in economic assessments, (see *e.g.* Stern (2006) and subsequent commentaries) and the equity weighting of spatially differentiated impacts (see *e.g.* Pearce *et al.* (1996) and subsequent commentaries). Other environmental health hazards, such as nano-particles, with potential long-term and/or hereditary impacts, are likely to raise similar technical and ethical challenges.

127. One might conclude from much of the review above that the gap between policy needs and the supply of empirical research findings in environmental health valuation – manifest in the uncertainties surrounding the existing WTP estimates – is likely to preclude a rapid increase in the up-take in such values for policy-analytical purposes in the near future. Certainly, a reduction in the extent of uncertainties resulting from new research is likely to be a relatively slow, incremental, process. However, policy-makers should also be aware that other data, including abatement/mitigation cost data, may be as uncertain, making other policy evaluation decision rules such as cost-effectiveness analysis similarly vulnerable. Additionally, as Rabl *et al.* (2005) demonstrate, it is also the case that if abatement costs are similarly uncertain – as they are in many environmental policy contexts – the net social costs of errors associated with cost-benefit outcomes are constrained by the fact that uncertainties in the estimation of individual components to some extent cancel each other out. Policy-makers may therefore be able to be more adventurous in their applications of environmental health values without fear of being wrong, or at least any more wrong than they would have been otherwise.

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ANNEX I: RECENT STUDIES THAT HAVE VALUED NON-FATAL HEALTH IMPACTS OF AIR AND WATER POLLUTION

Author(s) / Date / Policy input	***Pollutant***	***Source***	***Methodology***	***Health Measurement***	***Valuation***
Alberini, A *et al.* – 1997, Yes?	Various	N/S	Empirical – CV	Value of Reduced Morbidity (Taiwan)	262.58 million (USD 1992)
Bateman, IJ *et al.* – 2002, No	Various	N/S	Empirical – CV	WTP for impact reduction schemes (impact on human health)	Mean= 74.74 (GBP ??)
Bateman *et al.* – 2005, No	Ozone depletion (UV)	N/S	Empirical - CV	WTP for tax (public good) to avoid effects of UV	New Zealand 144.8 (£ 1998) England 58.7 (£ 1998) Scotland 41.9 (£ 1998) Portugal 16.4 (£ 1998)
Carlsson, F *et al.* – 2000, Yes	Various	Transport	Empirical – CV	WTP for Increased Air Quality, Per month	156 (SK 1996))
Chestnut, LG *et al.* – 1997, Yes	PM	N/S	Empirical - CV	WTP to avoid [1]: Symptom Day, RAD, WLD	16-63 (USD?) – Thailand 11-189 (USD?) – US
Chilton *et al.* – 2004, Yes	Various	N/S	Empirical - CV	Annual WTP to avoid one admission per household Hospital admission Annual WTP top avoid 2-3 days pa per household Breathing discomfort	 35.65 (GBP 2002?) 34.90 (GBP 2002?)
Desvousges, WH *et al.* – 1997, Yes	Various	N/S	Empirical – CA	WTP pa to avoid one day of respiratory or cardiac illness	Mild = 69-1443 (CD 1996) Severe = 683-1816 (CD 1996)
Dickie *et al.* – 2004, Yes	Various	N/S	Empirical - CV	WTP to relieve acute illness	Parent 75 (USD 2000) Child 160 (USD 2000)
Dickie, M *et al.* – 2004, Yes	Various	N/S	Empirical – CA, CV	WTP for Medication to avoid adverse health effects from air pollution, pa	Parent = 72-227 (USD ?) Child = 91-414 (USD ?)
Dzielgielewska *et al.* – 2005, No	PM	N/S	Empirical - CV	WTP for air quality improvements (Meeting EU standards)[1]: Acute bronchitis Asthma Minor ailments	 19 (USD 2000) 19 (USD 2000) 7.9 (USD 2000)
Hagen, DA *et al.* – 1999, Yes	Mercury	Energy Waste	Empirical – CV	 Reduction in Mercury Deposition (Minnesota)	 212 million (USD 1996)
Halvorsen, B – 1996, Yes	Various	Transport	Empirical - CV	Reduction in Health Risk	278 – 1133 (NOK 1993)
Hammitt *et al.* – 2005, No	Various	N/S	Empirical - CV	WTP to avoid adverse effects of air pollution in China : Cold Chronic bronchitis	 6-51 (USD 1999) 1500 – 3350 (USD 1999)
Ibanez, AM *et al.* – 2001, No	Various	N/S	Empirical – CV	WTP to avoid Acute Respiratory Illness episode	21.80-25.70 (USD 2000)
Muller, RA *et al.* – 2001, Yes	Various	N/S	Empirical – CV	WTP for 1/3 Health Improvement	84.2 (CD 1997)
Pearce, D – 1996, Yes	Various	N/S	Meta- analysis	Morbidity Costs Total Health Costs (5 Developing Countries)	82-472 million (USD?) 220-3138 million (USD?)
Ready *et al.* – 2004, Yes	General	N/S	Meta-analysis - CV	Morbidity - WTP avoid an illness episode[1]	GBP (?)

				Hospital Casualty Bed Cough Eyes Stomach	306 158 97 27 35 35
Rozan, A – 2004, Yes	Various	N/S	Empirical - CV	WTP for improvements in air quality (Strasbourg)	306-512 (FF?)
Saelensminde, K – 1999, No	Various	Transport	Empirical - SP	Valuation for local air pollution per percentage point change, per year, per household	127 – 255 (NOK 1993)
Yoo, SH *et al.* – 2001, Yes	Ozone	N/S	Empirical - CV	WTP for ozone pollution control	20 (USD 1997)

Notes [1] – see table 2 for valuations for specific health outcomes.

Authors/year	Pollutant	Source	Methodology	Measurement	Valuation
Barton (1999a), No	N/S	Effluent	Empirical - CV	WTP for improved water quality (Costa Rica)	Non significant
Barton (1999b), No	Sewage	Effluent	Empirical - CV	WTP for avoiding 1 day of illness episode (Costa Rica and Portugal) Eyes, Gastro-intestinal, Cough (USD 1999)	Eyes: Portugal – 91.62, Costa Rica – 50.67 Gastro-intestinal: Portugal 138.90, Costa Rica – 65.30 Cough: Portugal – 75.15 Costa Rica – 36.54
Bateman *et al.* (2006), No	N/S	Urban run-off	Empirical – CR, CV	WTP for river quality improvement (household pa) (GBP 1999)	12.07
Brox, J. A. *et al.* (2003), No	Sewage	Urban run-off	Empirical - CV	WTP for improved water quality (CD – 1994)	4.56-8.92
Burton, M. *et al.* (2001), No	GMO's	Agriculture	Empirical - CV	Purchasing behaviour	No Values
Crutchfield, S. R *et al.* (1997), Yes	Nitrates	Agriculture (Fertilisers)	Empirical -CV	WTP for improved drinking water per month per household (USD 1994)	50.31 – 60.93
Cuyno, L.C.M *et al.* (2001), Yes?	Pesticides	Agriculture (Onions)	Empirical - CV	WTP to avoid risks to health, per season (USD)	17
Echessah, P.N *et al.* (1997), Yes	Tsetse flies	Tsetse control	Empirical - CV	WTP per month for trapping programme (Kenyan Shillings 1993)	14.7
Foster, V & Mourato, S. (1997), No	Pesticides	Bread (Wheat)	Empirical - CR	WTP in pence per loaf per household for reductions in cases of ill health (GBP 1997)	0.0005 – 0.015 (or 240 per case of ill health)
Foster, V. *et al.* (1998), No	Pesticides	Agriculture (Wheat)	Empirical - CV	WTP for reductions per case of ill health, per loaf (GBP 1996)	0.73 – 1.05
Georgiou *et al.* (1998), No	Sewage	Effluent	Empirical -CV	WTP for reduction in risk of illness from quality of sea bathing water, per individual per annum	9.3 – 14.3 (GBP 1995)
Georgiou *et al.* (2000), Yes	Sewage	Effluent	Empirical - CV	WTP for new EC standard, per household, per annum (Sea bathing water) (GBP 1997)	42.29
Groothuiis, P. A *et al.* (1998), No	Toxic leachate	Waste (landfill)	Empirical (CV	WTA to allow hazardous waste facility per household (USD – 1998?)	1054
Hardner, J.J (1996), No	Various Micro-organisms	Human and animal waste	Empirical - CV	Ecuador; WTP for potable water (USD 1996 per household)	3.86
Hayne, T. P (1996), No	Toxic leachates	Waste (Landfill)	Empirical - CV	WTP for landfill cleanup per household, pa Value of lost income through illness (cancer) (USD 1996)	123.67 253039
Hoehn, J.P.,& Randall. (2002),No	Toxic leachates	Mining	Empirical - CR	WTP to reduce health risks (USD – 2002?)	127.3
Huang, J. *et al.* (2001), Yes	Pesticides	Agriculture (Rice/China))	Empirical - CV	Average health costs per farm (Yuan 1998)	9-36 (Yuan) or 1.08-4.34 (USD)
Jerrett, M.L (1996), Yes	Various	Various	Empirical	Defensive expenditures per capita in Ontario (USD – 1991)	347 – 912
Kwak, S-J. *et al.* (2004), Yes	Waste agricultural	Agriculture	Empirical - CV	WTP for recycling of WAF, per household, pa (Korean Won, 2001)	4641-5730

	film (WAF)				
Kuroyama, K *et al.* (2002), No	Oil	Accidental spills	Empirical - RP/SP	WTP to clean up oil spills (health effects) (Yen/100 person 2000)	74-287
Mourato, S *et al.* (2000), Yes	Pesticides	Agriculture (Wheat)	Empirical - CR	WTP for average 'green loaf' (organic - reduced risk to health and bird species). Per loaf; (GBP 1996)	1.37
Mourato, S. S. *et al.* (2003), Yes	Micro-organisms	Human and animal waste	Empirical - CA	WTP for reducing risk of stomach upset per household, pa. (GBP 2002)	1.1-2.0
Ozdemiroglu, E. J *et al.* (2004), No	Sewage	Sewage overflow	Empirical – CV and CE	WTP for improvement (GBP) per household, pa	58.94
Soderquist, T (1995), Yes	Radon radiation	Buildings (??)	Empirical - HP	WTP for reduction in radon concentration, per household, one time; (SK 1991)	21300
Soto Montes de Oca *et al.* (2003), No	N/S	N/S	Empirical - CV	WTP for improvement in water quality (Mexico) (pesos/bimonthly, 2001)	284
Travisi, C. M *et al.* (2004), No	Pesticides	Agriculture	Meta-analysis	WTP for risk reduction (for farmers) (USD 2000)	262
Travisi, C. M & Nijkamp, P. (2004), No	Pesticides	Agriculture	Empirical – CV, CCE	WTP to protect human health, per household, per month; (Euro 2003)	2.5- 3.14
Von Stackelberg, K. & Hammitt, J. (2005), No	PCB	NS	Empirical - CV	WTP to reduce risk (USD – 2005)	9.94 – 328.38