OECD Environment Working Papers No. 35

Policy Interventions to Address Health Impacts Associated with Air Pollution, Unsafe Water Supply and Sanitation, and Hazardous Chemicals

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https://dx.doi.org/10.1787/5kg9qx8dsx43-en

OECD

Organisation de Coopération et de Développement Économiques
Organisation for Economic Co-operation and Development

21-Jun-2011

English - Or. English

ENVIRONMENT DIRECTORATE

ENV/WKP(2011)5
Unclassified

Cancels & replaces the same document of 20 June 2011

ENVIRONMENT WORKING PAPER NO. 35
POLICY INTERVENTIONS TO ADDRESS HEALTH IMPACTS ASSOCIATED WITH AIR POLLUTION, UNSAFE WATER SUPPLY AND SANITATION, AND HAZARDOUS CHEMICALS

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JEL Classification: D04, Q25, Q51, Q52, Q53
Keywords: value of statistical life, meta-analysis, policy-implication, environment, health, transport

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JT03304335

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FOREWORD

This report was prepared by Dr. Alistair Hunt of the University of Bath, to support the elaboration of the Health and Environment chapter of the upcoming OECD Environmental Outlook to 2050.

The document is developed based on an analysis of recent peer-reviewed literature, and focuses on studies which value the impacts of human exposure to outdoor air pollution, unsafe water supply and sanitation and hazardous chemicals, and the costs of policy interventions to address such impacts. Based on this review, the report describes the degree to which conclusions can be drawn, with confidence, about the effectiveness of these policy interventions.

ABSTRACT

The purpose of the paper is to review the recent empirical literature relating to the quantification and valuation of the human health impacts of air pollution, hazardous chemicals, and unsafe water and sanitation, and their use in cost-benefit analysis, as an input to environmental policy decision-making. For each of these three environmental hazards, the nature and range of these health impacts are identified. The extent to which these impacts can, and have been, quantified and valued in monetary terms, is described. The use of this data in public policy-centred CBA is evaluated.

The health impacts associated with particulates and low-level ozone, and quantified on the basis of epidemiological evidence, ranges from minor respiratory conditions to cardio-pulmonary related mortality. CBA that includes these impacts is an established feature of air quality regulation formulation in North America and Europe. Indeed, reduced mortality impacts have dominated the benefits included in many recent appraisals of such policy development, though the robust valuation of these impacts is still evolving.

Heavy metals are associated with a variety of cancer impacts as well as on neurological development, renal dysfunction and a number of other impacts. These impacts are increasing being quantified and valued, in response to the requirement for more rigour arising from regulatory agencies in OECD countries. Increasingly sophisticated approaches to deal with the current attendant uncertainties are also being utilised.

The health risks from unsafe water and sanitation derive from faecal contamination, bacteria and viruses and include gastroenteritis, diarrhoea amongst others. OECD countries use of quantified information relating to these risks in CBA is currently in the context of wastewater management, where benefit-cost ratios are sometimes found to be less than one. In contrast, in developing countries, health risks are included in the appraisal of water supply investments. In these countries, health risks are significant but generally not as important as time savings in the benefits side of the CBA, though the quantification and monetisation of health risks is often rather partial.

JEL codes: D04, Q25, Q51, Q52, Q53.
Keywords: value of statistical life, meta-analysis, policy-implication, environment, health, transport

RÉSUMÉ

Ce rapport passe en revue les études empiriques récentes concernant la quantification et l'évaluation des impacts sur la santé humaine de la pollution de l'air, des produits chimiques dangereux, de l'insalubrité de l'eau et du défaut d'assainissement, et il examine l'utilisation des résultats dans les analyses coûts-bénéfices en vue d'étayer les décisions de politique environnementale. La nature et l'ampleur des impacts pour la santé sont identifiées pour chacun de ces trois dangers environnementaux. Ce document étudie aussi dans quelle mesure ces impacts peuvent être quantifiés et évalués en termes monétaires, voire le sont déjà. Enfin, il évalue l'utilisation de ces données dans les analyses coûts-bénéfices axées sur les politiques publiques.

Les effets sur la santé de l'exposition aux particules et à l'ozone troposphérique, quantifiés à l'aide de données épidémiologiques, vont de problèmes respiratoires mineurs à des décès par maladies cardio-pulmonaires. Les analyses coûts-bénéfices intégrant ces impacts constituent désormais une étape importante dans la formulation des réglementations en Amérique du Nord et en Europe. L'incidence d'une réduction de la mortalité occupe une place prépondérante parmi les avantages recensés dans un grand nombre d'évaluations récentes de cette évolution des politiques, même si la robustesse de l'évaluation de ces impacts doit encore évoluer.

Les métaux lourds sont associés à divers effets sur la santé tels que cancers, troubles du développement neurologique, dysfonctionnements rénaux et autres. Ces effets sont de plus en plus souvent quantifiés et évalués pour répondre aux exigences de rigueur accrue émanant des organismes réglementaires des pays de l'OCDE. Il est également fait appel à des approches de plus en plus élaborées pour gérer les incertitudes inhérentes à ces évaluations.

Parmi les risques pour la santé liés à l'insalubrité de l'eau et au défaut d'assainissement et émanant d'une contamination fécale, de bactéries ou de virus figurent, entre autres troubles, les gastro-entérites et les diarrhées. Les pays de l'OCDE utilisent des données chiffrées sur ces risques dans des analyses coûts-bénéfices portant sur la gestion des eaux usées, où le rapport bénéfices-coûts est parfois inférieur à un. En revanche, dans les pays en développement, les risques pour la santé sont pris en compte pour évaluer les investissements dans la distribution d'eau. Dans ces pays, les risques pour la santé sont importants, mais en général les bénéfices que représentent les économies de temps procurées par l'amélioration de la distribution d'eau le sont plus encore, même si la quantification et l'évaluation monétaire des risques sanitaires sont souvent relativement partielles.

Classification JEL : D04, Q25, Q51, Q52, Q53.
Mots-clefs : valeur d'une vie statistique, méta-analyse, implications politiques, environnement, transport, santé

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EXECUTIVE SUMMARY

The purpose of this paper is to review the recent empirical literature relating to the human health impacts of negative environmental externalities from air pollution, hazardous chemicals, and unsafe water and sanitation, and their use in cost-benefit analysis. More specifically, the objectives are to:

- Assess the nature and range of these health impacts;
- Identify how these health impacts can be expressed in monetary terms and used in policy-relevant applications;
- Understand the ways in which these impacts might differentially affect OECD and non-OECD countries; and
- Identify any conclusions that can be drawn from the current research on effectiveness of different policy interventions.

The range of health impacts

PM and ozone

In relation to air pollution, the specific focus here is on Particulate Matter (PM) and ozone. Whilst clearly there are other harmful emissions to air, many are effectively precursors of PM and ozone, and the former can now be seen, with some confidence, to be the most damaging to human health in terms of overall health damage costs, particularly in the longer term.

An extensive literature base supports the Concentration-Response (C-R) function linking PM with adverse health impacts, and there is now a general consensus that there is considerable strength of evidence for a causal link between long-term exposure to $PM_{2.5}$ and mortality. The picture regarding ozone is slightly more complicated, especially as it is difficult to disentangle the effects of ozone from the effects of PM. However, both evidence relating to short-term exposure, and the epidemiological evidence, are highly suggestive that ozone directly or indirectly contributes to cardio-pulmonary related mortality. Table E-1 summarises the range of the human health impacts from PM and ozone that have been identified, and those that have so far been valued in the research.

Table E-1: Quantified and un-quantified health impacts of $PM_{2.5}$ and ozone

Pollutant	Health endpoints – Quantified	Health endpoints – Un-quantified
PM	- Premature mortality - Bronchitis: chronic and acute - Hospital admission: respiratory, cardiovascular and cerebro-vascular - Emergency room visits for asthma - Cancer (Lung, trachea) - Lower and upper respiratory illness - Restricted activity days (adult) - Minor restricted activity days (Adult) - Work loss days - Asthma exacerbation (asthmatics) - Chronic cough (Child) - Cough (Asthmatic child) - Infant mortality	- Sub-chronic bronchitis cases - Bronchodilator usage - Low birth weight - Chronic respiratory disease other than chronic bronchitis - Non-asthma related emergency room visits - UVb exposure

Ozone	- Premature mortality: short term exposures - Hospital admissions – respiratory - Emergency room visits for asthma - Minor restricted activity days - School loss days - Asthma attacks - Acute respiratory symptoms	- Cardiovascular emergency room visits - Chronic respiratory damage - Premature aging of lungs - Non-asthma respiratory emergency room visits - UVb exposure

Source: Sources: Hunt and Ferguson (2010); EPA (2008).

Chemicals

The impact of chemicals on health has been the focus of significant research. Scientific evidence of the linkage between exposure and health impact usually consists of animal studies and some epidemiological studies of workers exposed to high concentrations, which has implications for the application of results to the analysis of policies affecting exposures of the general public to toxic chemicals.

Fairly robust epidemiological links exist for a range of exposures to heavy metals in particular and health endpoints. These include the following and are further summarised in Table E-2:

- Arsenic exposure and skin, lung and bladder cancers, cardiovascular mortality and still births;
- Cadmium exposure and osteoporosis and renal dysfunction;
- Chromium exposures and lung cancer;
- Lead exposure and impacts on IQ in children and anaemia;
- Mercury exposure and impacts on IQ in children, effects on the central nervous system and renal dysfunction; and
- Nickel exposure on lung cancer.

Table E-2: Main health impacts linked with exposure to heavy metals*

Pollutant	Health endpoint (relative severity of impact)	Route of exposure
Arsenic	Skin cancer (1.5)	Ingestion/inhalation
	Lung cancer (1)	Inhalation
	Bladder cancer (1.5)	Ingestion/inhalation
	Cardiovascular mortality (1)	Ingestion
		Inhalation
	Still birth/adverse pregnancy outcome (2)	Ingestion/inhalation
Cadmium	Osteoporosis (2)	Ingestion/inhalation
	Renal dysfunction (2.5)	Ingestion/inhalation
Chromium VI**	Lung cancer (1)	Inhalation
Lead	Children's IQ	Ingestion
		Inhalation
	Anaemia (2.5)	Ingestion
		Inhalation
	Cardiovascular illness	Ingestion/inhalation
Mercury	Children's IQ	Ingested methyl mercury
	CNS effects in adults – ataxia (2)	Ingested methyl mercury
		Inhaled Hg vapour
	Renal dysfunction – preclinical effects (3)	Inhaled Hg vapour
Nickel	Lung cancer (1)***	Inhalation

* Note that health impacts resulting from exposure to these substances are subject to exposure thresholds being reached.
** Chromium VI accounts for a relatively small proportion of total airborne chromium.
*** implied 3-fold difference in risk between ingestion and inhalation seems unlikely.
Source: Searle (2005)

Unsafe water and sanitation

Whilst epidemiological research is investigating the precise nature of the linkages between environmental risk factors and health impacts, attribution between risk factors is complex. In particular, the range of possible alternative pathways by which diseases are transmitted makes attribution difficult. For example, human and animal excreta can affect human health in the form of a number of different diseases through drinking water, sewage, indirect contact, and food through various pathways. It is, however, established that these fecal-oral diseases comprise the majority of the disease burden resulting from unsafe water and sanitation (WSS). See table E-3 for a summary. The vast majority of this disease burden is borne by lower-income countries.

Table E-3: Health Impacts associated with unsafe water supply and sanitation

Pollutant	Source	Health Impact
Faecal contamination	Bathing waters	Gastroenteritis, acute respiratory disease, infections, diarrhoea
Bacteria (protozoa)	Drinking water	Diarrhoea, amoebic dysentery, cholera, cryptosporidiosis
Viruses	Drinking water	Diarrhoea, gastroenteritis, meningitis, non-specific febrile illnesses

Valuation of health impacts

PM and ozone

A number of studies have derived unit values to capture the willingness-to-pay (WTP) to avoid these health impacts. There remains significant uncertainty in these WTP estimates, as indicated by the ranges shown in Table E-4. Mortality values are highest, followed by chronic bronchitis and hospital admissions. Non-OECD and older studies tend to estimate lower values for RAD and MRAD. When the costs of these impacts are examined in terms of total, per annum welfare costs, mortality and chronic bronchitis again represent the highest damage cost, with restricted activity days representing the next highest damages. From a sample of WTP values derived from such primary studies, there is a high degree of convergence across both OECD and non-OECD countries around valuations for the milder health impacts.

Table E-4: Summary of per incidence costs of health impacts associated with PM and ozone

Health Impact	Range of costs per incidence (USD, 2010 ppp)
Acute bronchitis	453-512
Chronic bronchitis	170 000-500 000
Respiratory hospital admissions	2 000-24 000
Cardiac hospital admissions	200-29 000
Asthma symptom day	38-54
Asthma attacks	75-280
Restricted activity day	30-150
Minor restricted activity day	38-53
Respiratory symptom day	6-50
Emergency room visit	80-670
Work loss day	80-150

Hazardous chemicals

In terms of valuation, some forms of cancer have been studied more than others – and there is considerable variation shown in terms of the values, given different contexts and cancer types. A summary of the main welfare costs associated with chemical-related health effects is given below. It can be seen that leukaemia is generally considered to have a significantly higher welfare impact than lung cancer, with skin cancer having the lowest impact in general. Neuro-developmental disorders can be valued at approximately $10,000 per case. It should be noted that care should be taken in the use of values in the table below for

policy evaluation – as specific cancer impacts of chemical releases may have acute or latent impacts. The issue of the valuation of cancer cases in children is also controversial.

Table E-5: Summary of welfare costs associated with chemical-related health impacts

	Medical treatment costs	Productivity loss costs	Dis-utility (value of a case)	Total WTP
Cancer (Lung)	11 000 (4 600 – 27 800)	70 000 (27 000 – 273 000)	400 000 (15 000 – 2 500 000)	481 000 (46 600 – 2,800 800)
Skin cancer	1 300 (125 – 9,300)	7000	1 000 (200 – 1,600)	9 300 (7 325 – 17 900)
Leukaemia	150 000 (60 000 – 250 000)	8000	2 500 000 (1.3m – 4.5m)	2,658m (1.368m – 4.758m)
Neuro-devt. Disorders	2 414 (428 – 4 400)	7,500 (2,500 – 18 000)	-	10,000 (3 000 – 22 000)

Note: Central values; ranges in brackets

Unsafe water and sanitation

The treatment of health differs a) to the extent that the specific health condition is identified and valued, and; b) according to the component of welfare costs that are addressed. For example, some studies measure welfare changes with respect to an overall change in the risk of illness from the pollution source – coastal bathing waters and urban run-off respectively. Other studies, however, identify WTP to avoid cases of specific illnesses associated with water pollution. With respect to (b), the studies divide between those that estimate the WTP to avoid illness and the pain and suffering implied, and those studies that estimate the direct economic costs from lost productivity and expenditure on medical treatment. Since the total welfare costs of a health impact are generally assumed to be the sum of the costs of illness and the WTP to avoid the pain and suffering, it is clear that these estimates are currently incomplete. However, it is also not possible simply to sum those estimates that we have identified because they have not been estimated for a common illness type. The following table gives a flavour of the range of values by pollutant. It should, however, be noted that the ranges of valuations are based on a small number of studies.

Table E-6: Estimated valuations of WTP for improved water quality

Willingness to pay:	Valuation (USD)
WTP to avoid an incidence of gastrointestinal illness	40-170
WTP per annum for improvements in water quality (OECD)	21-72 (per household) 14-21 (per individual)
WTP per annum for improvements in water quality (non-OECD)	13-260 (per household)

Policy interventions and cost-benefit analyses

PM and ozone

In most OECD countries, policy interventions in relation to air pollution have become increasingly integrated over the last 10-15 years. Examples include the Clean Air Act (USA and Canada), Clean Air for Europe, Air NEPM (Australia), all of which have set standards for air quality, focussing on target setting in relation to a range of air pollutants. These overall frameworks encompass a number of programmes of legislation targeting specific sectors, such as power generation, transport, industrial and domestic. In non-OECD developing countries, there are fewer examples of cohesive programmes for controlling air pollution. Currently, much of the focus in these countries is on specific policies for controlling emissions from transport.

The majority of the studies in this area originate from North America or Europe. There are a number of *ex ante*, policy-relevant analyses aimed at quantifying the health benefits of air pollution legislation. In

these cases, the purpose is likely to be political persuasion, and/or *post hoc* validation of legislation, rather than prompting allocation of financial (and other) resources.

Overall health benefits are dominated by the incidence avoided of premature mortality; the order of magnitude of costs changes very significantly between morbidity and mortality. *Ex post* analyses of the costs and benefits of legislation have often found both the *ex post* actual costs and benefits of compliance to be lower than those estimated *ex ante*.

Many of the studies in non-OECD countries emanate from international institutions, such as the World Bank or the World Health Organisation, and are designed to prompt policy choice and action. Reports by such institutions as the World Bank have noted that much of the burden of disease from air pollution is borne by developing countries and arises from road transport emissions. In China, for example, health damage costs are estimated at between 1.2 and 3.8% GDP.

No *ex ante* or *ex post* cost-benefit analyses were found for non-OECD countries. However, there were some studies estimating (*ex ante*) the benefits of introducing air quality policies, and these all identified very significant benefits in reduced health damage costs, from USD 10's of millions at city-wide level to USD billions at country level.

Many of the cost-benefit analyses available are regulatory impact studies. There are a number of studies that summarise the potential costs and benefits of reaching air quality targets across USA, Canada and EU. A notable result from a number of studies is that net benefits in relation to ozone control tend to be negative, given that the costs, in the short term at least, are very high. This finding has been replicated in recent cost-benefit analyses/comparisons of a number of different policies. However, US EPA notes that the Clean Air Act requires the EPA to set standards to protect human health regardless of economic factors. Studies specifically focusing on pollution from transport also demonstrate a high level of net benefits. Some studies report cost to benefits ratios, which from a policy-making perspective is probably a helpful metric for choosing between different policy options, but it is often not possible to make this assessment.

Three recent studies make an integrated comparison of the costs and benefits of a range of policy measures at country or regional level (ICGB UK, 2007; US EPA, 2010; EU EEA, 2010).

IGCB (2007) evaluated the impacts of selected air quality policies in the road transport and electricity supply industries. In general, the selected policies were in line with various European directives for these sectors, but with some additional national policies, such as road pricing, emission zones and incentive packages. A number of policy measures relating to control of emissions from transport (road and marine) had positive cost-benefit values. Other measures had negative values at the lower end of the range, but positive at the upper end of the range, and these related to the implementation of even more stringent emissions control, and were therefore likely to have a longer latency period before the benefits are felt. Measures relating to the phasing-out of older vehicles and policies relating to management of domestic consumption and emissions, showed negative net present values, and are therefore, according to this analysis, less preferable as policy options. The annual present values of benefits to health from PM reductions were consistently positive across all policy variants; however, the annual present value of benefits to health from ozone is negative in many, if not most policy variants.

EEA (2010) looked at the impact of selected policy measures on Europe's air quality. The focus was on policies relating to control of emissions from transport and from energy, and the relevant European air pollutant policy framework was the EU National Emission Ceilings (NEC) Directive (EC, 2001b), which imposes ceilings to be met by 2010. Within this there are sector-specific emission reduction measures – Euro standards for road vehicles (e.g. EC 2007), the EU Large Combustion Plant (LCP) Directive (EC,

2001a) and the EU Integrated Pollution Control (IPPC) Directive (EC, 1996). The study report reports in terms of percentage reduction in health impact from road transport policies and industrial combustion policies: *For $PM_{2.5}$*, the reduction of health impact (in Years of Life-years Lost, YOLL) from road transport policies was 13%, whilst for industrial combustion sector policies it was 60% (averaged across all EU countries). For *ozone*, the reduction in health impact (YOLL) from road transport policies was 17% (averaged across all EEA countries), and for industrial combustion policies, YOLL increased by 17%. The health impacts from ozone will vary significantly across EU countries as a result of the policies, some countries experiencing positive health impacts, and some negative health impacts, such that when averaged across the whole EU it produces an overall increase in YOLL.

The US EPA study of the benefits and costs of the 1990 Clean Air Act Amendments reports on the additional abatement policies introduced as a consequence of these amendments. The study estimated total life years gained in 2020 to be 1 900 000. From a cost-benefit perspective, it estimates an overall benefit-to-cost ratio of approximately 28:1

For the present and for the next decade or two, the value (net present) of policies targeting reduction of emissions from road traffic is most obvious. Given the increasing congestion in many of the growing mega-cities in developing countries, continuing to target road transport emissions reduction would seem an obvious priority.

Whilst premature mortality is clearly the greatest economic cost arising from outdoor air pollution, policies that target reductions in this health impact may not be the most cost efficient and effective policy intervention for developing countries where resources and incentives may be differently aligned to the ways in which they are in developed economies. From the perspective of a developing economy, it *may* be a more useful step in the legislative process to focus resources on reducing the morbidity that stretches local health services, and where the benefits are more demonstrable in the shorter- to medium-term.

Ozone emission reduction policies carry a high cost, and the benefits are likely to be felt only in the longer term. Furthermore, at country or regional level, the effects of ozone vary dramatically and thus the benefits are not experienced uniformly, or even positively, across the whole policy-affected area. However, overall, damages from air pollution policies implemented in US, Canada and Europe are demonstrably reducing and so it is not difficult to demonstrate return on investment.

Regarding co-benefits of climate change policies, GHG reductions affect climate change in the long run, whereas benefits of reducing local air pollution are likely to be felt in the shorter- to medium-term. This works the other way around as well, in that targets to reduce local air pollution are likely to have a positive impact in relation to *climate change*. However, there are clearly some trade-offs involved that would need to be better understood and quantified.

Hazardous chemicals

Policy interventions relevant to hazardous chemicals have taken the following approaches:

- Targeting specific chemicals (mainly the heavy metals identified above) in specific sectors. Examples include legislation to reduce mercury emissions from power generation, or to reduce the negative impacts of the use of lead in paint;
- Targeting a number of hazardous chemicals in a specific sector. An example of this is EC legislation of chemicals used in the toy production sector;
- Developing an overarching approach to the monitoring and regulation of a wide range of chemicals – an example is the European REACH programme, around which there has been a significant amount of cost-benefit analysis.

A review of studies that have considered health as part of detailed cost-benefit analysis shows a number of major issues. These include:

- The treatment of ancillary impacts of chemical regulations is limited. Actions to mitigate chemical releases are likely to have impacts on other pollutants, and these should be considered where possible.
- Values applied for health impacts vary: there is marked variation in the treatment of latency, the cancer premium and the treatment of age. There is also sometimes inconsistency in the values applied for health endpoints and the nature of the endpoint.

The presentation of quantification methods and studies used to derive values is rather mixed. Variations in assumptions such as discounting affect the comparability of results.

Valuation of morbidity endpoints in the analysis of chemicals policy is limited.

Sensitivity analysis has been based around the use of upper and lower values, and "best guess" values. This may be because of the time frame involved in conducting these analyses, which are often driven by regulatory timetables that are quite short. Advanced quantitative analysis of uncertainties using *e.g.* Monte Carlo methods is seldom conducted.

Unquantified health impacts are sometimes used to justify policies with significant costs – *e.g.* in EC (2008), cost-benefit analysis is used to justify increased regulations on the production of toys, with the most stringent regulation being proposed despite a cost of over $13 billion. Given the costs, further research on the unquantified health impacts – even using simple expert judgement or Delphi methods to do some quantification may have been justified.

Unsafe water and sanitation

The range of policy interventions to reduce pollution from water and wastewater are well established, and include the following categories of intervention: Providing access to safe water and sanitation, including wastewater collection and transportation – aimed at reducing incidence of diseases, especially waterborne and water-washed diseases; Investing downstream in wastewater treatment for safe disposal and reuse – to accrue additional health benefits, including those from improved quality of recreational waters; Investing upstream in managing the supply/demand balance sustainably – aimed at increasing quality of life due to reliable water supply.

The studies identified range from assessments of water supply and waste treatment at the municipal level to those at the world regional level. The varying geographical scales reflect the fact that resource allocation is determined at these different scales.

There are some indications that for developing countries, investment in WSS options produces largely favourable results in terms of benefit-cost ratios (BCR), and that the limiting factor in determining such investment in developing countries will be the absolute levels of financing available.

In more developed countries, the findings are rather different in that BCR's of less than 1 have sometimes been found. This may be because the benefit/cost scenarios can be more complicated. For example, investments in drinking water and sewage cannot be considered in isolation from (upstream) resource protection and (downstream) wastewater treatment. A noteworthy study looked at a number of WSS options in different combinations, finding that if policy makers were to invest in cholera vaccinations *before* implementing water interventions, the economic outcomes would be more positive.

Health impacts are typically found to be a significant, but not dominant, parameter in the determination of the BCR; time savings are more important in the benefits of the majority of WSS options in developing countries.

The coverage of health impacts in the majority of the studies is limited to consideration of diarrhoea, fatal and non-fatal, and it is clear that valuation of these end-points is partial. For instance, all non-fatal cases of diarrhoea are valued on the basis of the costs of illness; they do not include a WTP estimate for the pain and suffering component of the welfare cost that one would expect to be considerably greater than the COI component. In the case of valuing fatal cases of diarrhoea, the studies either use a non-WTP method based on lost lifetime earnings or use a value of statistical life derived from a single study undertaken in Bangladesh that is below the levels derived in the majority of studies undertaken globally. For these reasons, it may be expected that the health impacts are considerably under-represented in CBAs of WSS to date.

POLICY INTERVENTIONS TO ADDRESS HEALTH IMPACTS ASSOCIATED WITH AIR POLLUTION, UNSAFE WATER SUPPLY AND SANITATION, AND HAZARDOUS CHEMICALS

Introduction

1. This paper examines the relationships between negative environmental externalities and health. More specifically, it focuses on the health impacts resulting from Particulate Matter (PM) and ozone arising from outdoor air pollution, unsafe water and sanitation, and hazardous chemicals. Key objectives for the paper are to use the available literature to:

- Assess the nature and range of these health impacts;
- Identify how these health impacts are expressed in monetary terms and used in policy relevant applications;
- Understand the ways in which these impacts might differentially affect OECD and non-OECD countries; and
- Identify any conclusions that can be drawn from the current research on effectiveness of different policy interventions

2. The following is organised in four sections. The first section summarises the literature in relation to air pollution and human health. It begins by contextualising the literature on the health impacts associated with exposure to PM and ozone. The policy-relevant literature is then examined, with specific focus on recent cost-benefit studies, and conclusions drawn as far as possible, about effectiveness of different interventions. The section concludes with an overview about methodological and conceptual uncertainties inherent in valuing health impacts. Section Two follows the same pattern for hazardous chemicals, while Section Three focuses on unsafe water and sanitation. Although we highlight in broad terms the range of health effects associated with these pollutant themes, the subsequent focus is on the health effects that have been monetised and used in cost-benefit analyses (CBA).

3. The focus on CBA here does not imply that other decision tools such as cost-effectiveness analysis are not useful in these policy contexts; rather the paper looks to identify the extent to which the potential of CBA has been exploited and the extent to which its use may be limited.

4. CBA can provide important input for priority setting and decision making in environmental policy, taking *i.a.* impacts related to human health into account. There are, however, uncertainties *e.g.* in the assessments of primary environmental impacts, in the assessments of the epidemiological consequences for human health of these impacts, and in the economic valuation of the relevant impacts. CBA estimates should therefore be applied cautiously, and include sensitivity analyses of important parameters.

5. Furthermore, net present values of expected benefits and expected costs are not the only relevant inputs in decision-making. Assessments of distributional impacts, and assessments of low-probability, high-impact outcomes, based on a precautionary principle, are, for example, also important.

6. The review focuses on studies that have been published (a) in peer-reviewed journals, identified via a number of search engines including ScienceDirect, IngentaConnect, SpringerLink and PubMed, and databases including EVRI and EconLit; or (b) on Government or international institution websites whose databases were searched using their own search engines. Relevant working party or conference papers were included.

7. The review focuses on studies published within the last 15 years, thereby excluding those from 1995 and before. It includes both studies from OECD countries and from non-OECD countries – mainly, but not exclusively BRIC (Brazil, Russia, China and India) countries. As can be concluded from the foregoing, the review is representative and illustrative rather than exhaustive. It is clear, however, that the majority of the literature that reports on the quantification and monetisation of health impacts of environmental pollution has – to date – been in the context of air pollution and its regulation. As a consequence, more space is given to this body of literature in this report.

1. Outdoor air pollution and human health

1.1 Health impacts associated with PM and ozone

8. Whilst other areas of human welfare impacts associated with PM and ozone have been valued, this paper focuses specifically on human health impacts, the range of which is summarised in Table 1. It is clearly the case that children and the elderly (and those with other underlying health issues) are more vulnerable to the effects of air pollution that other segments of the population. Whilst some attention has been given to the issue of how VSL might vary with age (see above), the issue of children's health is more complicated, both from an epidemiological and a valuation perspective.

9. WHO (2004) conducted a review of the epidemiological and toxicological literature in this area and concluded that there was sufficient evidence to suggest a causal link between outdoor air pollution and a number of health outcomes for European children. Such outcomes included respiratory deaths in the post-neonatal period, lowered birth weight, adverse effects on lung development and function, asthma and asthma aggravation, increased cough and bronchitis and enhanced allergic sensitisation. The evidence also suggested that many of the morbidity and mortality effects related to air pollution occur via an interaction with respiratory infections, which are relatively more frequent among young children. WHO noted that whilst the relative risk estimates for these health outcomes were generally small, the amount of ill health amongst children attributable to air pollution is high. They also noted that the mechanisms through which these effects occur were not yet well understood and that there was a need for more epidemiological research.

Table 1: Quantified and un-quantified health impacts of $PM_{2.5}$ and ozone

Pollutant	Health endpoints – Quantified	Health endpoints – Un-quantified
PM	- Premature mortality - Bronchitis: chronic and acute - Hospital admission: respiratory, cardiovascular and cerebro-vascular - Emergency room visits for asthma - Cancer (Lung, trachea) - Lower and upper respiratory illness - Restricted activity days (Adult) - Minor restricted activity days (Adult) - Work loss days - Asthma exacerbation (Asthmatics) - Chronic cough (Child) - Cough (Asthmatic child) - Infant mortality	- Sub-chronic bronchitis cases - Bronchodilator usage - Low birth weight - Chronic respiratory disease other than chronic bronchitis - Non-asthma related emergency room visits - UVb exposure

Ozone	- Premature mortality: short term exposures - Hospital admissions – respiratory - Emergency room visits for asthma - Minor restricted activity days - School loss days - Asthma attacks - Acute respiratory symptoms	- Cardiovascular emergency room visits - Chronic respiratory damage - Premature aging of lungs - Non-asthma respiratory emergency room visits - UVb exposure

Sources: Hunt and Ferguson (2010); EPA (2008).

10. An extensive literature base supports the Concentration-Response (C-R) function linking particulate matter with adverse health impacts. Most of the current economic analyses involving particulate matter and morbidity and mortality rely on a small number of high-quality studies (Abbey *et al*., 1995; Pope *et al*., 2002; Laden *et al*., 2006; EPA, 2006). In the UK, The Committee on the Medical Effects of Air Pollutants (COMEAP, 2006) reviewed the available literature and concluded that there was considerable strength of evidence for a causal link between long-term exposure to $PM_{2.5}$ and mortality. They also concluded that the American Cancer Society study (Pope *et al*., 2002) provided the best single source of information for quantifying the effects of PM. They advised that the appropriate coefficient (expressed in terms of percentage change in relative risk of all-cause mortality per $\mu m/m^3$ change in annual average $PM_{2.5}$) to use at that time was 1.06 (95% CI: 1.02-1.11). More recently, US EPA (2010) drew a similar conclusion and used the same coefficient in its analyses for the study '*The Benefits and Costs of the Clean Air Act*'. So too did the European Environment Agency (2010) in investigating the impact of selected policy measures on Europe's air quality ('*Impact of selected policy measures on Europe's air quality*').

11. The picture regarding ozone is more complicated, since it is difficult to disentangle the effects of ozone from the effects of PM. The EPA in its Final Regulatory Impact Analysis for Ozone NAAQS, citing the Ozone Criteria Document (EPA, 2006) notes that "We were not able to separately quantify all of the PM and ozone health effects that have been reported in the ozone and PM criteria documents in this analysis for four reasons: (1) the possibility of double counting (such as hospital admissions for specific respiratory diseases); (2) uncertainties in applying effect relationships that are based on clinical studies to the potentially affected population; (3) the lack of an established concentration-response relationship; or (4) the inability to appropriately value the effect (for example, changes in forced expiratory volume) in economic terms". However, both evidence relating to short-term exposure, and the epidemiological evidence, are highly suggestive that ozone directly or indirectly contributes to cardio-pulmonary related mortality. Time-series studies, including the National Morbidity, Mortality and Air Pollution Study (NMMAPS) and meta-analyses of these studies (Bell *et al.*, 2005; Ito *et al.*, 2005; and Levy *et al.*, 2005) have not been conclusive, although the 2006 EPA Criteria Document concludes that "the results from these meta-analyses, as well as several single- and multiple-city studies, indicate that co-pollutants generally do not appear to substantially confound the association between ozone and mortality" (p. 7-103)". The most commonly used coefficient for ozone is 1.003 (95% CI: 1.001-1.0043), from the Bell *et al.* (2004) NMMAPS data.

1.2 Valuation of specific health impacts

12. Table 1 showed the range of health impacts that have been quantified. A number of studies have derived monetary values to capture the welfare effects associated with these health impacts. These valuations are derived from empirical studies of a specified population's willingness-to-pay (WTP) to avoid health impacts. Table 2 below gives a sample of WTP values derived from such primary studies; these values are for avoiding one case of each of the health outcomes listed. It should be noted that the presentation of central WTP values produced by the individual studies hides the fact there is significant uncertainty attached to these estimates, arising from outstanding issues linked to methodological practice and statistical representativeness. Methodological uncertainties are outlined below.

13. An examination of the values estimated reveals – unsurprisingly – that premature death, or mortality, is the health outcome which people would pay most to avoid. Of non-fatal (morbidity) health outcomes, chronic bronchitis associated with PM is the most costly in terms of cost per incidence, with values ranging from around USD 170 000 to USD 500 000. Hospital admissions for either respiratory or cardiac conditions carry the second highest cost per incidence, with values ranging from USD 2000 to USD 15 000. There is a fairly high degree of convergence around valuations for restricted activity days (RAD) and minor restricted activity days (MRAD), the former ranging from about USD 64 to USD 130 in more recent studies, and the latter from USD 38 to USD 52. Non-OECD and older studies (*e.g*., Chestnut *et al*., 1997; Hubbell *et al*., 1999; Strukova *et al*., 2006) tend to estimate lower values for RAD and MRAD.

14. When the costs of these impacts are examined in terms of total, per annum, welfare costs in a country or region resulting from air pollution, mortality and chronic bronchitis represent the highest damage cost, with restricted activity days representing the next highest damages. Hospital admissions (respiratory and cardiac) also constitute a significant damage cost (see *e.g.* Holland *et al.*, 2005).

Table 2: Willingness-to-pay valuations of health impacts from primary studies

Location/ date	**Health impact valued**	**Valuation per capita/case** 2010 USD, PPP corrected
11 EU countries; Desaigues *et al.* (2007)	Life year	40,000
3 EU countries, Alberini *et al.* (2006)	Prevented fatality	1 million
Kerala; Baby et al., (2009)	Symptom day	40
Thailand and USA; Chestnut *et al.,* (1997)	Symptom day Reduced activity day Work loss day	Thailand= 25; USA= 20 Thailand= 38; USA= 32 Thailand= 70; USA= 120
China; Hammitt *et al.,* (2005)	Cold Chronic Bronchitis	8-60 1750-3700
Bogota, Colombia; Ibanez *et al*., (2001)	Acute respiratory illness	30
UK; Chilton *et al*., (2004)	1 hospital admission 2-3 days breathing discomfort	35 Per household 35 Per household
Toronto; Chestnut *et al*., (1997)	One day cardiac or respiratory illness	Mild =90-1700 Severe= 850-2150
USA; Dickie *et al*., (2004)	Acute illness	Parent=80 Child= 170
Hattiesburg, USA; Dickie *et al*.., (2004)	Medication to avoid adverse health effects of pollution	Parent= 78-240 Child= 100-440
Vigo, Spain; Dubourg *et al*., (2001)	Bed Hospital Cough ER Eyes	40 160 1 53 8
Poland; Dzielgielewska *et al*., (2005)	Acute bronchitis Asthma Minor ailments	22 22 10
EU 15; Navrud *et al.,* (2001)	Hospital ER Bed Cough Eyes Stomach	5000 150 52 6 6 52

15. There is a relatively high degree of convergence across both OECD and non-OECD countries around valuations for the milder health impacts (*i.e.* symptom day, reduced activity day, cold, acute respiratory illness, 2-3 days breathing discomfort), where a range of USD 10 to USD 50 captures the ranges of WTP means, although given the range of different definitions of health impacts, it is difficult to

be precise. This convergence across regions is supported by the findings of Chestnut *et al.* (1997) who in fact find WTP values for symptom days and RADs to be marginally higher in Thailand than in USA.

16. As indicated above, there remain a number of unresolved methodological issues in valuing morbidity and mortality that result in a significant degree of uncertainty in the robustness of the estimates. A number of recent papers (*e.g.*, Navrud, 2008; Hunt, 2008; Lindhjem *et al*., 2010; Chestnut and De Civita, 2009) outline these methodological differences in some depth. We briefly summarise three of the outstanding issues below. These issues are common to the health valuations undertaken or applied in the WSS and hazardous chemicals contexts; therefore we do not repeat this discussion in the sections of the report that address these risk contexts.

1. Use of non-market valuation techniques to value the dis-utility component of welfare change.

17. How reliable are values derived from revealed preference and stated preference valuation techniques – generally, and in the health context? For example, are people able to distinguish between small differences in mortality risk and differentiate their preferences accordingly (scope tests)?

18. Health valuations are normally derived by either Stated Preference or Revealed Preference techniques. The former are based on methods which explicitly ask individuals how much they are willing to pay (WTP), or to accept, in compensation for a small change in risk. The practical difficulties associated with this technique are that it calls for very complex survey design, which can often place a very high burden on the cognitive capacity of survey respondents (Hunt and Ferguson, 2010). For example, the extent to which individuals can distinguish between levels of risk is still uncertain. In their recent meta-analysis and review of the literature, Lindhjem and Navrud (2010) observed that people's WTP was not sufficiently sensitive to the size of risk change to give fully reliable results. Alberini *et al.* (OECD VERHI project, 2010) also note that risk increment discrimination is cognitively very difficult for survey respondents, and that this may be further compounded by a focus on the perception of risk identified.

19. Revealed Preference (RP) methods are based on hedonic wage approaches which use market prices to value risk. Whilst this data is clearly more easily obtained, there are problems with it. In valuations relating to health, wage premiums for risky occupations are normally, although not exclusively, used. The limitations of these studies are that they generally only draw on a small subset of the population (for example, the working age segment, or the segment of population financially able to move house in response to an environmental risk), and that they reveal immediate perceptions of risk, rather than building in the actual and latent risk associated with environmental hazards. Thus, RP methods are likely to overstate valuations, in the case of labour markets prices, (see for example Bayer *et al.*, 2009).

2. How should premature mortality impacts be valued? For example, when should Value of Statistical Life (VSL) and Value of a Life Year (VOLY) metrics be used?

20. The recent literature defines two options for valuation of mortality impacts; the Value of Statistical Life (VSL), and the value of a life year (VOLY). Krupnick (2004) in his edited peer review of the methodology proposed for the Cost Benefit Analysis of the Clean Air For Europe (CAFE) programme (Holland, 2004) notes that, unlike the VSLs, which are computed from estimates of the WTP for risk reductions using hedonic wage (HW) or stated preference surveys, VOLY estimates have generally been computed from a VSL estimate, usually from a HW study. The strength of the VOLY measure is that it is able to deal with age differences. (As noted above, most deaths due to environmental policy inaction would be to elderly, and it is sometimes argued that to treat elderly and non-elderly as equivalent for valuation purposes is inappropriate because so many fewer life-years are lost when elderly die.) However, Chestnut (2009) in a review of the literature concluded that the evidence around variation in WTP with age is inconclusive, and that none of the empirical studies found WTP to decline consistently with age. Navrud

(2008) concludes that where VOLYs are used, they should be computed directly from stated preference studies, rather than indirectly from VSLs.

21. There is, arguably, an emerging consensus that Years of Life Lost (YOLL) is a useful, or necessary, metric for cost-benefit analyses. Papers on methodology advice emerging from the EEA, the EPA and the Australian Environment Agency all recommend its use. Further, in the health sector, YLL are sometimes coupled with an assessment of the quality of the life year, following the QALY (Quality Adjusted Life Year) approach (Hammitt, 2002). However, Hubbell (2002) experimented with the use of QALY's in evaluating air pollution policy, but concluded that there were a number of ethical and methodological problems with the approach.

22. Very few studies have estimated WTP for VOLY directly. Chilton *et al.* (2004), for Defra, performed a CV survey of gains in life expectancy and constructed a "best" estimate for VOLY of USD 32 000. Desaigues *et al.* (2007) improved on the DEFRA CV survey instrument and performed the same CV survey in 9 European countries. The estimated VOLY varied between countries, but the sample size for each country is small and they recommend using only estimates separate for EU-15 (plus Switzerland) and the new Member States at EUR 41 000 and EUR 33 000; respectively; and a weighted (by population) EU-25 average value of EUR 40 000.

23. In terms of current usage for policy evaluation purposes, EC DG Environment recommends using a VSL value of USD 1.6 million; US EPA recommends the use of USD 7.4 million and Chestnut *et al.* recommend USD 6 million, whilst a recent meta-analysis suggests a central value of USD-2005 2.9 million for OECD countries (Navrud *et al*., 2011). In terms of VSL's for non-OECD countries, recent work conducted for OECD gives guidance on best practice in the transfer of VSL estimates from one country to another (Biausque, 2010; Lindhjem *et al.*, 2010, Navrud *et al*., 2011). One might also expect that VSL will vary with GDP, the logical result of which will be lower VSLs in many developing countries. The ethical issues inherent in this conclusion may be more appropriately resolved through a political process.

3. Aggregation and equity: For example, should we use the same value for children and adults? Or is there a risk of double-counting due to altruism when parents asked to value children?

24. The literature suggests that the WTP that parents express to avoid a given health impact to their children is greater than that to avoid the same impact on themselves (see *e.g.* Hunt and Ortiz, 2006). Much of the research relating to children and air pollution has been carried out by Dickie and team (see Dickie and Ulery, 2002; Dickie and Brent, 2002; Dickie and Hubbell, 2004; and Dickie and Messman, 2004). 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.

25. However, there are a number of methodological and philosophical issues in achieving reliable values on which to base WTP estimates. It is, for example, difficult to disentangle the type of altruism that is at play in CV studies. 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. A recent study by the OECD (2010a) aimed to address a number of the issues relating valuing children's health, by conducting a 3-country European study, using different stated preference methods. Their results generally confirmed earlier studies in that it showed that parents have stronger preferences for reducing mortality risk to their children than to themselves. Across the UK and the Czech Republic, for example, parents are willing to pay from two thirds to twice more for their children

than they are for themselves. These results are clearly significant from a policy perspective, given that they have a strong bearing on VSL.

1.3 Health damage costs of air pollution

26. A number of studies have attempted to monetise a range of total health costs resulting from air pollution, presumably as a spur to legislative action. Table 3 shows a sample of the results from a range of such studies.

Table 3: Overall damage costs from outdoor air pollution

Location and study	Health impact	Per annum valuation (USD million, 2010ppp)
3 Cities (Taiwan), (Alberini *et al.*, 1997)	Morbidity	350
USA, (EU EPA 2002)	Morbidity	230
5 developing countries, (Pearce *et al.*, 1996)	Morbidity Costs	130 – 700
Vigo Spain, Dubourg *et al.*, 2001	Morbidity	1.8- 4
Ukraine 19 cities (Strukova *et al.*, 2006)	Morbidity (PM)	180
Canada, (McCubbin *et al.*, 1999)	Morbidity	100
EU27, (EU, EEA, 2010)	Total health damages (PM & Ozone)	200,000 - 630,000
EU, (EU EEA, 2006)	Total health damages	230,000 – 700,000
USA, (Bloyd *et al.*, 2002)	Total health damages	55,000 – 110,000
Singapore, (Euston *et al.*, 2003)	Total health damages (PM)	4,500 2,200= mortality 2,300= morbidity
USA, (McCubbin *et al.*,1999)	Total health damages (due to anthropogenic pollution)	1,300,000
USA, (US EPA 1996)	Total health damages	180,000-550,000
USA, (Muller *et al.*, 2007)	Total health damages	25,000
5 developing countries, (Pearce *et al.*, 1996)	Total health damages	300 – 4,400
Madrid, Spain, Monzon *et al.*, 2004	Total health damages	500
India, (Srivasta et al 2002)	Total health damages	700
China (30 cities), (Wei *et al.*, 2009)	Total health damages	10,000
Sao Paulo, Brazil, (Miraglia, 2005)	Total health damages for children and elderly	4
Guangdong province, China, (Zhou *et al.*, 2005)	Total health damages (PM)	230
China, (World Bank, 2007)	Total Health damages	1.2-3.8% of GDP
Thailand, (World Bank, 2002)	Total health damages – PM (6 cities)	850
Tehran Province, Iran, (Karimzadegan *et al.*, 2007)	Total health damages	600
Indonesia, (World Bank, 2003)	Total health damages	600

27. This selection illustrates that the majority of the studies in this area originate from North America or Europe. Many (although not all) of the studies in non-OECD countries emanate from international institutions, such as the World Bank or the World Health Organisation, and are designed to prompt policy choice and action. Reports by such institutions (*e.g.* Cohen *et al.* for the World Bank) have noted that much of the burden of disease from air pollution is borne by developing countries and arises from road transport emissions. Thus, whilst the results shown might suggest that overall damage costs are lower in non-OECD countries, this is misleading, since these costs are likely to be a larger percentage of that country's GDP. In China, for example, environmental health damage costs are estimated at between 1.2 and 3.8% GDP (World Bank, 2007).

28. This summary of overall damage costs also reinforces that the overall health costs are dominated by the cost of premature mortality; the order of magnitude of costs changes very significantly between

morbidity and mortality. For example, Pearce *et al.* (1996) estimate morbidity costs to constitute 15-45% of total costs, implying that the remainder (55-85%) are due to mortality impacts.

29. There is also an indication from the table that over the past two decades, damage costs have been reduced in the EU and USA, presumably as policies for emissions reduction take effect. Illustrating this, the totals estimated in Muller *et al.* (2007) are 10% of the totals estimated by USEPA (1996) a decade earlier.

Benefit studies

30. Table 4 below summarises a number of illustrative studies from OECD member countries which generally fall into the category of *ex ante*, policy relevant analyses aimed at quantifying either the health benefits of air pollution policies, and in these cases, the purpose is understood to be political persuasion, and/or *post hoc* validation, rather than a desire for *ex ante* efficient allocation of financial, and other, resources.

31. A small number of studies identifying *ex ante* policy benefit emanating from developing countries were found. The following are illustrative. Anas *et al.* (2009) compared the effectiveness of different policy instruments to reduce traffic congestion and CO_2 emissions in Beijing. The study showed that a congestion toll is more efficient than a fuel tax in reducing traffic congestion, whereas a fuel tax was more effective as a policy instrument for reducing gasoline consumption and emissions. Improvements in car efficiency were also found to reduce congestion, fuel consumption and CO_2 emissions significantly. Such a policy clearly benefits wealthier households that own a car. It may also benefit those too poor to own a car, but who benefit from improved health as a result of the reduced traffic-related air pollution.

32. Cesar *et al.* (2002) estimated the benefits of reducing PM and ozone emissions in Mexico City and concluded that the annual benefits of achieving a 10% or a 20% reduction in PM_{10} and ozone emissions would be about USD 760 million and USD 1.49 billion respectively. They did not, however, identify and evaluate specific policy options; rather, they position the paper as providing the motivation to do so.

33. Stevens *et al.* (2005) estimated the benefits of retrofitting particulate filters to road vehicles in Mexico City. They found the annual health benefits to be of the magnitude of USD 0.41 - 0.58 million in 2005 and USD 0.48 - 8.1 million in 2010, reflecting uncertainty in the benefit quantification process.

34. Larson *et al.* (1999) evaluated six emissions control options for reducing pollution from two industrial facilities in Volgograd, Russia, chosen to illustrate a cost-benefit analysis. The options, 3 for one facility and 3 for the other, were a variety of emissions and dust control measures. They estimated the monetised health benefits from 5 of the 6 options to generate a net benefit in terms of reduced mortality of USD 40 million. These studies give some clear guidance around the estimated monetary benefits of policy intervention to reduce air pollution from PM and ozone. The majority of the benefit (as previously noted) is from reduced mortality.

Table 4: Ex ante quantified benefits of specific or general policies

Location/ date	Policy	Valuation/annum (USD million, 2010ppp)
South Appalachian Mountains, USA, Abt associates, 2002	SAMI emission controls	Mortality = 45 000 Chronic Bronchitis = 2 000
EU 15, AEA Technology, 1998	Air quality targets for trophospheric ozone	Chronic bronchitis = 1 600 - 1 800 RAD's=500-580
UK, UK Dept of Health, 1999	Reductions in ozone and PM	PM = 600 Ozone = 400
EU 15, Holland *et al*., 1999	Emissions ceilings for atmospheric pollutants	Morbidity and mortality = 300 - 35 000
USA, Hubell *et al.*, 1999	Emissions reductions from HGV in 2030	Total health = 80 000 Mortality = 75 000
USA, Hubbell *et al*. ,2005	Benefits of achieving US ozone standard	Total benefit over 3 years = 6 200
USA, US EPA, 2002	Meeting NESHAP standards (Industrial boilers)	20 000 - 22 000
China, Brajer *et al*., 2004	China and WHO clean air standards	80 000
Mexico City, Cesar *et al.*, 2005	Reduction in ozone, 2010	10% reduction = 1 100 20% reduction = 2 100

Cost-benefit studies

35. Many of the cost-benefit analyses that have been undertaken are regulatory impact studies – that is, studies designed to inform and persuade stakeholders prior to the implementation of regulation. In many OECD member countries such impact studies will form the focus of consultation activity prior to regulation being enforced.

36. A sample of these studies are summarised in Table 5. There are a number of studies that summarise the potential costs and benefits of reaching air quality targets, such as those done in USA, Canada and EU. The vast majority of studies find net benefits from the air quality regulations considered. The potential net benefits are of comparable orders of magnitude. US EPA (2005), estimating for both USA and Canada, found the benefits from reducing emissions from industrial combustion to be around USD 116 billion, whereas for Canada alone, net benefits could be around USD 5 billion (Coyle *et al*., 2003). An early European study (EU, 1999), estimated the annual net benefits for achieving air quality targets at USD 65 - 75 billion. A notable result from a small number of studies (*e.g.* US EPA, 2008) is that net benefits in relation to ozone control tend to be negative, given that the emission reduction costs, in the short term at least, are very high.

37. The studies in Table 5 that focus on regulation of pollution from transport also demonstrate a high level of net benefits.

Table 5: OECD Countries - Selected sample of Cost-benefit analyses (NPV and B-C ratios)

Location / date	Policy	NPV (USD million (M) or billion (B) 2010 PPP in **bold;** B-CRs
USA & Canada 2005	Control of emissions from electricity generation	**130 B**
Canada 2003	Reaching Canada Wide Standards for air quality (from industrial combustion)	**3000 M**
UK 1999	Reductions in PM and ozone	**PM = 650 M** **Ozone = 400 M**
EU 15 1999	Reaching EC 2010 air quality targets	**6000 - 7000**
USA – 29 states + DC 2004	Interstate air quality rule – general cap and trade program reducing emission from utilities and transport	**60 B (2010); 85 B (2015)** 2.9 (2010) 3.7 (2015)
USA 2005	Evaluation of US Acid Rain Program	**130 B** 3 (2015)
USA 2008	Regulatory Impact analysis for Ozone National Ambient Air Quality Standards	**- 4.3 to - 11 B**
USA 2008	Regulatory impact analysis relating to control of emissions from diesel marine engines	**3.4 B**
Mexico City 2005	Retrofit of particulate filters on vehicles	**Net benefit 0.1-1.3 M**
Mexico City 2005	Analysis of emission control policy options (road transport)	Benefit cost ratios - **Reducing LPG leaks most favourable**
USA 2010	Regulatory impact analysis of GHG and fuel emission reductions- heavy duty vehicles	**Net benefit = 1.76 B**

Cost and benefit analyses of a range of policy measures.

38. Three recent studies (ICGB UK, 2007; US EPA, 2010; EU EEA, 2010) were identified which evaluated the impacts of selected policy measures at country or regional level. Two further studies (EU EEA, 2006; OECD, 2009) looked at the ways in which policies relating to climate change might have co-benefits in terms of air pollution, and these are summarised in the following paragraphs.

39. IGCB (2007) evaluated the impacts in the UK of selected air quality policies in the road transport and electricity supply industries. In general, the selected policies were in line with various European directives for these sectors, but with some additional national policies, such as road pricing, emission zones and incentive packages. Approaches used included the full Impact Pathway approach (see Markandya *et al.*, 2004) for some policies, but for others, benefits were assessed on the basis of emission reductions only. Health impacts were monetised using a damage-cost approach. The specific policy measures evaluated are identified in Table 6.

Table 6: Summary of policy measures evaluated in "An Economic Analysis to inform the Air Quality Strategy". ICGB (2007)

Measure	Description
A -	Euro 5/VI NO_x and SO_x reductions in new Large Goods Vehicles (LGVs) and Heavy Goods Vehicles (HGVs)
A2	Euro 5/6/VI – revised scenario Greater NO_x and SO_x reductions in new LDVs and HDV's
B	New Euro 5/6/VI – high intensity Greater NO_x and SO_x reductions in new LDV's and HDV's
C	Programme of incentives for early uptake of Euro 5/V/VI (relating to Measure A)
C2	Programme of incentives for early uptake of Euro 5/6NI/VI standards (relates to Measure A2)
D[a]	Programme of incentives to phase-out most polluting vehicles
E	Programme of incentives to increase penetration of low-emission vehicles
F	Impact of national road pricing scheme
G[b]	Low emission zones (divided into 3 sub-measures)
H[c]	Retrofit particulate filters on HGV's and captive fleets
I[d]	Domestic consumption: switch from coal to natural gas or oil
J	Domestic consumption: product standards for gas-fired appliances
K1, 2, 3	Large combustion plant measures (*e.g.* specifying emission control technologies)
L	Small combustion plants measure
M	Reducing national VOC emissions by 10%
N	Shipping-based measures (*e.g.* specifying emission control technologies)
O	Combined measures C+E
P	Combined measures C+L
Q	Combined measures C+E+L
R	Combined measures C2+E+N

Notes: [a] – Modelled over 5-10 years; [b] – modelled over 5-8 years; [c] – modelled over 13 years; [d] – modelled over 15 years.

40. Table 7 summarises the findings in terms of net present value of costs, and net present value.

Table 7: Summary of annual present value of costs and net present value of policies (2010 USD million)

Measure / costs (Annualised present value of costs)	Annualised net present value (brackets denotes -ve)
A / 400-420	80 - 801
A2/ 788 – 793	(264) - 539
B/ 983 – 1003	(432) - 514
C/ 409 – 417	148 - 947
C2/ 816 – 823	(246) – 595
D1/ 117	(101) – (96)
E/ 61	63 - 112
F [a]	
G (London only)/ 33 – 88	(107) – (13)
H1/ 68	(33) – (17)
I / 43	(23) – (15)
J / 196	(179) – (148)
K 2/ 273	(107) - 34
L/ 9	18 - 57
M/ 249	(249) – (248)
N/ 1	245 - 576
O/ 470 – 478	186 - 978
P/ 418 – 426	163 - 1000
Q/ 479 – 487	203 - 1053
R/ 878 – 885	33 - 1211

Notes: [a] – a high degree of uncertainty means that it is impossible to predict
(This table does not include the annual present value of benefits to crops and buildings.)

41. As can be seen in the table, a number of policies (A, C, E, L and N, and the combined measures O - R) have positive cost-benefit values. Most of these measures relate to control of emissions from transport (road and marine). Other measures have negative values at the lower end of the range, but are positive at the upper end of the range. These include A2, B, and K2, which relate to the implementation of increasingly stringent emissions controls for transport vehicles that are therefore likely to have a longer latency period before the benefits are felt. Measures D, G, H, I J, and M, relating to the phasing-out of older vehicles and policies relating to management of domestic consumption and emissions, show negative net present values, and are therefore, according to this analysis, not preferable as policy options. The studies show that the annual present value of benefits to health from PM reductions are consistently positive across all policy variants; however, the annual present value of benefits to health from measures addressing ozone emissions is negative in most policy variants.

42. ICGB (2005a) noted – contrary to the optimism bias literature – that for many, although not all, policies, the *ex post* implementation costs have been less than those predicted *ex ante*, a finding also confirmed by Harrington *et al.* (2000), who noted that *ex ante* estimates particularly overstated the costs for market-based programmes. ICGB noted that *ex ante* CBA may not adequately predict the impact of innovation on costs. At the same time, other evidence indicates that benefits may also be over-estimated, *ex ante*. The net effect of these two patterns is therefore that the average costs per unit of environmental improvement are proven to be relatively well estimated, *ex ante*.

43. EEA (2010) looked at the impact of selected policy measures on Europe's air quality. The focus was on policies relating to control of emissions from transport and from energy production. The relevant European air pollutant policy framework was the EU National Emission Ceilings (NEC) Directive (EC, 2001b). These regulations imposed emission ceilings to be met by 2010. Within this there are sector-specific emission reduction measures – Euro standards for road vehicles (*e.g.* EC 2007), the EU Large Combustion Plant (LCP) Directive (EC, 2001a) and the EU Integrated Pollution Control (IPPC) Directive (EC, 1996). The EU Air Quality Directive (EC, 2008) came into force in 2008, many of the target values or limits came into force in 2010.

44. The general approach to the analysis considered three technology scenarios:

- 'actual scenario' – which modelled developments in emissions and air quality trends based on measures actually introduced as a result of the policies;
- "no application' scenario – which modelled how emissions and air quality would have developed given no abatement measures over the period of the regulation;
- 'full application' scenario – which modelled how emissions and air quality would have developed given full application of all relevant legislation to all sources considered, with no time lag.

45. The study does not facilitate identifying the monetised impacts of specific policies but reports in terms of percentage reduction in health impacts – mortality and morbidity – from road transport policies vs. industrial combustion policies:

$PM_{2.5}$ actual *vs.* no application scenario – reduction of health impact (in YOLL):

46. Road transport policies = 13% reduction. A further improvement of up to 10% is predicted under the 'full application' scenario.

47. Industrial combustion sector policies actual *vs.* no application scenario = 60% reduction (averaged across all EU countries). 'Full application' could deliver further improvement of between 5 and 30%.

Ozone actual *vs.* no abatement reduction in health impact (YOLL)

48. Road transport policies = 17% reduction (averaged across al EEA countries). Further improvement under 'full application' predicted to be 10%

49. For industrial combustion policies = YOLL increased by 17%, due to changes in atmospheric chemistry composition, but under full application "a small reduction in YOLL due to ozone exposure is predicted".

50. US EPA, Office of Air and Radiation (US EPA, 2010) conducted a study of the benefits and costs of the 1990 Clean Air Act Amendments, and as with the IGCB study, the analyses were conducted on a 'with policy' and a 'without policy' basis. It reports on the totality of CAAA policies. The study estimated total life years gained (or reduction in years of life lost – YOLL) in 2020 to be 1 900 000. Table 8 summarises health benefit outcomes from the Amendments.

Table 8: Summary of monetised health benefits of the US Clean Air Act

Health impact	Pollutant	Benefits in 2020 (2010 USD M)
Mortality – adults > 30	PM	1 700 000
Mortality – Infants	PM	2 500
Mortality – all ages	Ozone	55 000
Chronic bronchitis	PM	35 000
Non-fatal Myocardial Infarction	PM	21 000
RHA	PM, Ozone	1 100
CHA	PM	200
ER visits – respiratory	PM, ozone	44
Acute Bronchitis	PM	94
Lower respiratory symptoms	PM	42
Upper respiratory symptoms	PM	60
Asthma exacerbation	PM	130
MRADs	PM, Ozone	6 700
Work loss days	PM	2 700
School loss days	Ozone	480
Outdoor worker productivity	Ozone	170
Total health benefits =		1 827 220

Source: US EPA (2010)

51. The overall benefit-to-cost ratio was estimated to be approximately 28:1

52. For comparison purposes, Table 9 summarises the benefit-to-cost ratios from previous EPA studies.

Table 9: Summary of studies monetising the benefits and costs of the Clean Air Act from its inception

Study	Benefits	Costs	B-C ratio
CAA 1970 through 1990, EPA retrospective study (USD 1990)	22.2 trillion	523 billion	42:1
CAAA 1990 through 2010, EPA prospective study (USD 1990)	690 billion	180 billion	4:1
Stratospheric ozone protection, EPA prospective study (USD 1990)	530 billion	27 billion	20:1

Source: Van Atten, C. & L. Hoffman-Andrews, 'The Clean Air Act's Economic Benefits; Past, Present and Future.' The Main Street Alliance, 2010.

53. Neither the US EPA nor the EEA studies describe analyses that would enable different policy interventions to be compared with one another in cost-benefit terms. This means that on the basis of the published reports, it is not possible to draw conclusions about the efficiency and effectiveness of specific policies. The ICGB study does, however, facilitate a comparison of this sort, and as noted above, it would seem that over the shorter term, at least, policy interventions relating to emissions reduction from transport have the greater net present values, thereby indicating that there is more scope to improve air quality through regulations in this sector than in stationary sources.

Co-benefits of climate change mitigation policies

54. Two large-scale studies have estimated the co-benefits of policies to reduce greenhouse gas (GHG) emissions (EU, 2006; OECD, 2009). The EU (2006) study concluded that action to combat climate change will deliver considerable ancillary benefits in air pollution abatement by 2030; in the order of USD 10 billion per year, leading to a reduction in damage to public health (*e.g.* more than 20 000 fewer premature deaths per year) and to ecosystems (See Table 10 for summary of benefits to human health). These ancillary benefits will be greater in 2030 than in 2020.

Table 10: EU estimated co-benefits of climate change mitigation policies

Year	Scenario	Life years lost due to $PM_{2.5}$ (millions)	Premature deaths ($PM_{2.5}$ and ozone (thousands)	Monetised health damage (EUR B)
2000		3.62	370	280-790
2030	EEA Baseline	2.64	311	210-650
	EEA Climate action	2.45	288	190-600
	EEA Climate action MFR	1.66	200	130-420

EEA Climate action – consistent with the target of limiting global temp change to 2 degrees C above pre-industrial level; EEA MFR a climate action scenario that assumes maximum feasible reductions for air pollutants
Source: EEA (2006). Notes – Baseline is the CAFÉ scenario.

55. EU (2006) concludes that significantly greater efforts will be necessary in the form of further targeted air pollution abatement measures in order to move closer to the EU long-term air quality objectives. Even if the maximum technically feasible land-based reduction measures for abatement of air pollution were combined with climate policies, the study projects that there will still be 200 000 annual premature deaths by 2030 from ozone and fine particles. Reductions in emissions from non-land based sources, especially shipping, are therefore necessary if the health impacts are to be further reduced.

56. Bollen *et al.* (2009) for OECD took a broader global approach to reviewing the co-benefits of climate change mitigation policies and found that whilst there was evidence for co-benefits to local air pollution control from climate change mitigation policies (policies focused on reduction of greenhouse gas emissions), and that these provide some incentives to participate in global climate change mitigation efforts, the magnitude of these co-benefits, and some of the trade-offs involved, have only been partially explored. Their analysis therefore aimed to assess the magnitude of the co-benefits of mitigation policies in terms of reduction in local air pollution and its implication for human health. The study found that reductions in GHG emissions induced large reductions in local air pollution emissions, with potentially significant positive impacts for human health. Modelling a scenario where GHG emissions were cut by 50% in 2050, air pollution related premature deaths in 2050 could be reduced by 20% – 40%, depending on region. More co-benefits accrued to OECD countries in the short run, but to non-OECD countries in the longer run, or under a scenario of less ambitious mitigation effort. Using a VSL of USD 1 million (2000 USD) for the European Union, the valuation of these health co-benefits in 2050 under the 50% cut scenario were estimated to vary from 0.7% GDP for the EU to 4.5% for China. (It should be noted that OECD currently recommend the use of a VSL of USD 3.5 million (2005 prices) for EU-wide policies for policy makers, thereby implying a possible trebling of these benefits). Bollen *et al.* (2009) found the optimal policy mix to be one which entailed a less than 50% GHG emissions reduction, but a stronger focus on

local air pollution control. The authors note that this finding is highly sensitive to regional VSL values as well as to discount rate assumptions. Thus, if the value for VSL was constant across OECD and non-OECD countries, further air pollution control would be implied, at the expense of GHG emission control.

1.4 Summary and conclusions

57. An extensive literature base supports the Concentration-Response (C-R) function linking PM with adverse health impacts, and there is now a general consensus that there is considerable strength of evidence for a causal link between long term exposure to $PM_{2.5}$ and mortality. The picture regarding ozone is more complicated since it is difficult to disentangle the effects of ozone from the effects of PM. However, both evidence relating to short-term exposure, and the epidemiological evidence, are highly suggestive that ozone directly or indirectly contributes to cardio-pulmonary related mortality.

58. A number of studies have derived unit values to capture the willingness-to-pay (WTP) to avoid these health impacts. There remains significant uncertainty in these WTP estimates, as indicated by the existing estimates. Mortality values are highest, followed by chronic bronchitis and hospital admissions. Non-OECD and older studies tend to estimate lower values for RAD and MRAD. When the costs of these impacts are examined in terms of total, per annum, welfare costs, mortality and chronic bronchitis again represent the highest damage cost, with restricted activity days representing the next highest damages. From a sample of WTP values derived from such primary studies, there is a growing degree of convergence across both OECD and non-OECD countries around valuations for the milder health impacts.

59. In most OECD countries, policy interventions in relation to air pollution have become increasingly integrated over the last 10-15 years. Examples include The Clean Air Act (USA and Canada), Clean Air for Europe, Air NEPM (Australia), all of which have set standards for air quality, focussing on target-setting in relation to a range of air pollutants. These overall frameworks encompass a number of programmes of legislation targeting specific sectors, such as power generation, transport, industrial and domestic. In non-OECD developing countries, there are fewer examples of cohesive programmes for controlling air pollution. Currently, much of the focus in these countries is on specific policies for controlling emissions from transport.

60. The majority of the studies in this area originate from North America or Europe. There are a number of *ex ante*, policy relevant analyses aimed at quantifying the health benefits of air pollution legislation. In these cases, the purpose is likely to be political persuasion, and/or *post hoc* validation of legislation, rather than the desire for the efficient allocation of financial (and other) resources.

61. Overall health benefits are dominated by the incidence avoided of premature mortality; the order of magnitude of costs changes very significantly between morbidity and mortality. *Ex post* analyses of the costs and benefits of legislation have often found both the *ex post* actual costs and benefits of compliance to be lower than those estimated *ex ante*.

62. Many of the studies in non-OECD countries emanate from international institutions such as the World Bank or the World Health Organisation and are designed to prompt policy choice and action. Reports by such institutions as World Bank have noted that much of the burden of disease from air pollution is borne by developing countries and arises from road transport emissions. In China, for example health damage costs are estimated at between 1.2 and 3.8% GDP.

63. No *ex ante* or *ex post* cost-benefit analyses were found for non-OECD countries. However, there were some studies estimating (*ex ante*) the benefits of introducing air quality policies, and these all identified very significant benefits in reduced health damage costs, from USD 10's of millions at a city-wide level to USD billions at country level.

64. Many of the cost-benefit analyses available are regulatory impact studies. There are a number of studies that summarise the potential costs and benefits of reaching air quality targets across USA, Canada and EU. A notable result from a number of studies is that net benefits in relation to ozone control tend to be negative, given that the costs, in the short term at least, are very high. This finding has been replicated in recent cost-benefit analyses/comparisons of a number of different policies. However, EPA notes that the Clean Air Act requires the EPA to set standards to protect human health regardless of economic factors. Studies specifically focusing on pollution from transport also demonstrate a high level of net benefits.

65. Three recent studies make an integrated comparison of the costs and benefits of a range of policy measures at country or regional level (ICGB UK, 2007; US EPA, 2010; EU EEA, 2010).

66. IGCB (2007) evaluated the impacts of selected air quality policies in the road transport and electricity supply industries. In general the selected policies were in line with various European directives for these sectors, but with some additional national policies, such as road pricing, emission zones and incentive packages. A number of policy measures relating to control of emissions from transport (road and marine) had positive cost-benefit values. Other measures had negative values at the lower end of the range, but positive at the upper end of the range, and these related to the implementation of even more stringent emissions control, and were therefore likely to have a longer latency period before the benefits are felt. Measures relating to the phasing-out of older vehicles and policies relating to management of domestic consumption and emissions, showed negative net present values, and are therefore, according to this analysis, less preferable as policy options. The annual present value of benefits to health from PM reductions were consistently positive across all policy variants; however, the annual present value of benefits to health from ozone is negative in many, if not most, policy variants.

67. EEA (2010) looked at the impact of selected policy measures on Europe's air quality. The focus was on policies relating to control of emissions from transport and from energy and the relevant European air pollutant policy framework was the EU National Emission Ceilings (NEC) Directive (EC, 2001b), which imposes ceilings to be met by 2010. Within this there are sector specific emission reduction measures – Euro standards for road vehicles (*e.g.* EC 2007), the EU Large Combustion Plant (LCP) Directive (EC, 2001a) and the EU Integrated Pollution Control (IPPC) Directive (EC, 1996). The study reports in terms of percentage reduction in health impact from road transport policies and industrial combustion policies: For $PM_{2.5}$, the reduction of health impact (in YOLL) from road transport policies was 13%, whilst for industrial combustion sector policies it was 60% (averaged across all EU countries). For *ozone*, the reduction in health impact (YOLL) from road transport policies was 17% (averaged across all EEA countries), and for industrial combustion policies, YOLL increased by 17%. The health impacts from ozone will vary significantly across EU countries as a result of the policies, some countries experiencing positive health impacts, and some negative health impacts, such that when averaged across the whole EU produces an overall increase in YOLL.

68. The US EPA study of the benefits and costs of the 1990 Clean Air Act Amendments reports on the additional abatement policies included in these amendments. The study estimated total life years gained in 2020 to be 1 900 000. From a cost-benefit perspective, it estimates an overall benefit-to-cost ratio of approximately 28:1.

69. It should be noted that the three studies outlined here have varying degrees of coverage of health impacts. Whilst the US EPA study is perhaps the most comprehensive, the European studies have more partial coverage. Similarly, the values used – particularly in relation to mortality – are only slowly moving to be more in line with each other, the US values being higher than those in Europe. Whilst this may to some extent reflect differences in preferences as a result of income differences, etc., it also appears to be the case that these also reflect differences of opinion regarding methodological issues, and the results of studies that use differing methodologies.

70. For the present and in the near future, the positive net present value found for policies targeting reduction of emissions from road traffic seems clear. Given the increasing congestion in many of the growing mega-cities in developing countries, continuing to target road transport emissions reduction would seem an obvious priority.

71. Ozone emission reduction policies are found to carry a high cost, and the benefits are likely to be felt only in the longer term. Furthermore, at country or regional level, the effects of ozone vary dramatically and thus the benefits are not experienced uniformly or even positively across the whole policy-affected area.

72. Regarding co-benefits of climate change policies, GHG emission reductions are projected to reduce climate change impacts in the long run, whereas benefits of reducing local air pollution are likely to be felt in the shorter- to medium-term. This works the other way around as well, in that measures to reduce local air pollution are likely to have a positive impact in relation to climate change. However, there are clearly some trade-offs involved that would need to be better understood and quantified.

2. Hazardous Chemicals

2.1 Health impacts associated with exposure to chemicals

73. The impacts of exposure to chemicals on health have been the focus of increased attention in recent years. There is a wide range of health outcomes associated with chemicals, including *e.g.* increased risk of cancer, disorders of the central nervous system and osteoporosis, dependent on the chemical.

74. Current evidence suggests that – as a result of their combined exposure patterns and toxicity – the metals with the highest risk potential are Arsenic (As), Cadmium (Cd), Chromium (Cr) (in oxidation state 6, designated as CrVI), Mercury (Hg), Nickel (Ni) and Lead (Pb). They have a variety of adverse health impacts, most prominently including cancers for As, Cd, CrVI and Ni, and neurotoxic impacts for Pb. The major impacts of Hg appear also to be neurotoxic, but their quantification still poses many problems at the present time. Among the health impacts of dioxins are endocrine disruption and cancers, but only the latter can be quantified at the present time. We also consider cancers due to inhalation of benzene, formaldehyde, butadiene and benzo(a)pyrene. Other metals and salts, such as manganese, thallium, uranium and vanadium, also have health impacts associated with exposure to them (see the US EPA IRIS database for details of specific health effects).[1]

75. The Concentration Risk Factors (CRFs) for cancers due to inhalation given by EPA are stated as unit risk factors (URF), defined as the probability, per $\mu g/m^3$ of ambient concentration, of getting a cancer due to a lifetime exposure (taken as 70 yr). With our definition of the CRF as impact for a 1 year exposure, the slope, sCR, is the unit risk divided by 70.

76. The scientific evidence usually consists of animal studies and some epidemiological studies of workers exposed to high concentrations. There are major methodological issues when using either occupational and/or animal studies for quantitative human risk assessment; see, for example, US EPA (1996) or HEI (1995). Issues to be considered include that:

- The reliability of risk estimates in occupational studies depends crucially on the reliability of estimated long-term exposures of the study subjects;
- Use for public health risk estimation requires extrapolation both to low concentrations and to possibly more susceptible individuals;

[1] www.epa.gov/iris/index.html.

- Quantitative use of risk estimates from animal studies may also involve low-dose extrapolation and quantitative animal-to-human scaling.

77. These difficulties have led to substantial diversity in the acceptance of quantified risk estimates for development of cancer.

78. For many of these pollutants, in particular dioxins and the most toxic metals (As, Cd, CrVI, Hg, Ni and Pb), the dose from ingestion of food is for most people about two orders of magnitude larger than the inhalation dose. However, the health impact per dose can be different depending on the intake mode: for example, according to current knowledge, Cd, CrVI and Ni are carcinogenic only via inhalation. For CRFs determined by epidemiological studies, the question arises whether the effect of the ingestion dose should be added to that of inhalation. This depends on what exactly was measured in the epidemiological study. Typically, the study population was exposed simultaneously via inhalation and ingestion. Even if the result of a study is stated as CRF, *i.e.* in terms of ambient air concentration, it may in fact reflect the total dose. If the ratio of inhalation and ingestion for the general population is different from that of the study population, one does not know how to apply the CRF unless one can make reasonable assumptions about the separate inhalation and ingestion doses of the study population and the relative effectiveness of these two dose routes.

79. For the carcinogenic metals, As, Cd, Cr-VI and Ni, the unit risk factor (URF) is shown in the third line of Table 11 and the CRF slope sCR in the fourth. At the present time, the evidence for cancers due to ingestion of Cd, Cr and Ni is not sufficiently convincing to indicate a DRF.

Table 11: CRFs and DRFs, per kg emitted, for the carcinogenic metals.

	As	Cd	Cr-VI	Ni
Inhalation				
URF [cancers/(pers·70yr·µg/m^3)]	4.30E-03	1.80E-03	1.20E-02	2.40E-04
sCR [cancers/(pers·yr·kg/m^3)]	6.14E+04	2.57E+04	1.71E+05	3.43E+03
Ingestion				
slope factor [cancers/(mg/(kgbody·day))]	1.50E+00			
sDR [cancers/kg]	1.07E+00			

Unit risk and slope factors from IRIS www.epa.gov/iris. concentration-response function (CRF)
sCR = slope of concentration-response function
sDR = slope of dose-response function.

Dioxins

80. For 'dioxins' (collective name for polychlorinated dibenzo-*p*-dioxins or PCDDs and polychlorinated dibenzofurans or PCDFs), there is only a dose-response function for the exposure route via ingestion. This dose-response function also applies to the dioxin-like substance group of polychlorinated biphenyls (PCBs). The term 'dioxins' refers to a group of polychlorinated, planar aromatic compounds with similar structures, chemical and physical properties. This group of compounds consists of 75 polychlorinated dibenzo-*p*-dioxins (PCDDs) and 135 polychlorinated dibenzofurans (PCDFs), of which 2,3,7,8-TCDD is the most toxic and most studied congener.[2]

81. Dioxins are not produced commercially and have no applications, other than for preparation of analytical standards and research materials. They are formed during any low temperature combustion

[2] www.besafenet.com/report.html#Executive%20Summary.

process in presence of precursor compounds containing carbon, oxygen, hydrogen and halogen atoms (Bumb *et al.*, 1980) such as cooking and burning coal for heat, or as unwanted by-products of industrial processes. It is, thus, evident that the primary sources of dioxins today are combustion processes (Fiedler *et al.*, 2000). The principal route by which dioxins are likely to result in health impacts is through ingestion of foodstuffs, especially dairy products, meat, fish and shellfish in which dioxins accumulate. Very low levels are also found in plants, water and air and pose a minimum threat to human health (Quaß *et al.*, 2000).

82. In contrast to the dioxins, PCBs have been produced intentionally. The marketing and use of PCBs has been very restricted in the EU through Directive 85/467/EC, and some European countries, as Sweden, had even banned the use of PCBs as early as 1973. An international convention, the POPs (Persistent Organic Pollutants) Convention, currently in negotiation, aims to ban the production and use of PCBs worldwide. However, today, PCBs are widely spread. They are largely present in transformers and capacitors where they were used as dielectric fluids, but also in building material, carbon-less copy paper, lubricants, surface coatings, adhesives, plasticisers, and inks among other uses.

83. Several human epidemiological studies and numerous studies in experimental animals have been carried out of dioxins. There can be acute as well as chronic effects. Dioxins cause changes in laboratory animals that may be associated with developmental and hormonal effects; however, the mechanism of carcinogenicity is unclear. Whether the biochemical changes may result in adverse health effects in people and at what concentrations is not very well known.

84. In laboratory experiments with animals, TCDD (tetrachlorodibenzo-p-dioxin) has been found to be one of the most potent toxins known, with LD50 ranging from 0.6 to 3000 μg per kg of body weight for different mammals (LD50 is the dose that kills half of a test group) (Tschirley, 1986). This wide range of values suggests that extrapolation from one animal species to another is quite uncertain.

85. The dioxins 2,3,7,8-TCDD and HxCDD are said by EPA to be "the most potent carcinogen(s) evaluated by the EPA's Carcinogen Assessment Group". The slope factor is 1.0E+06 cancers per (mg/(kgbody·day)) (EPA 2000).

Benzene

86. Benzene is a known human carcinogen. However, risk quantification is complicated by lack of quantitative data, short follow-up at low exposure concentrations, co-exposures to other potential carcinogens, and the fact that the body breaks down benzene to metabolites which seem to be more toxic than the parent substance. Individual variation in susceptibility or metabolism may therefore influence the risk at any given exposure. There are many occupational studies investigating exposure to benzene and development of cancer, especially leukaemia.

87. The US EPA risk assessment for benzene gave a unit risk factor of:

$8x10^{-6}$ cancers/(pers·70yr·μg/m^3) (US EPA, 1990).

88. Many different risk estimates have been derived, using different assumptions about the pattern of exposures, the shape of the CRF, and the extrapolation to low concentrations. These are similar to the estimates of Crump (1994) who gives a range of: 4.4 to $7.5x10^{-6}$ cancers/(pers·70yr·μg/m^3) for the URF of leukaemia.

1,3-Butadiene

89. 1,3-butadiene is potentially carcinogenic to both the white and red blood cell systems. Animal studies have shown that it is carcinogenic in mice and other rodents. There is, however, wide discrepancy

in metabolism between different species, complicating extrapolation to humans. Although the available animal evidence for 1,3-butadiene and comparison with substances of similar chemical structure would support the classification of butadiene as a human carcinogen, the available human data is limited.

90. 1,3-butadiene is a major ingredient of synthetic rubber and, being volatile, the route of absorption is primarily inhalation. The epidemiological evidence consists mostly of mortality studies which use qualitative estimates or exposure categories rather than estimates of actual lifetime exposures, and with limited consideration of other workplace exposures. There is no evidence available on cancer risks to the general population from ambient exposures. The human studies cannot be used directly in quantified risk assessment because sufficiently reliable estimates of past exposures are not available. The US EPA (1991) URF of $3x10^{-4}$ cancers/(pers·70yr·μg/m^3) is based on multi-stage modelling of animal (mice) experimental data. An updated estimate by RIVM (1994) of 0.7 to $1.7x10^{-5}$ is much lower. The contribution of this pollutant to the total damage cost of vehicle emissions is judged to be extremely small.

Polycyclic Aromatic Hydrocarbons (PAHs)

91. These are ring compounds resulting from the incomplete combustion of organic material and which jointly share carbon atoms. They cover a wide range of substances, including benzo[a]pyrene (BaP). The relationship between BaP and other PAHs differs for various types of emission but has been shown to be relatively similar in the ambient air of several towns and cities.

92. There is strong evidence, including from epidemiological studies (*e.g.* Redmond *et al*., 1976; Hurley *et al*., 1983; Armstrong *et al*., 1994), to suggest that certain components of PAHs, and specifically benzo[a]pyrene, are carcinogenic (lung cancer) in humans; and that nitroaromatics as a group pose a hazard to health. Benzo[a]pyrene specifically, rather than PAHs as a group, is labelled as a probable human carcinogen. As these compounds form complex mixtures and are also absorbed onto particulates, it is difficult to quantify levels of human exposure and so is difficult to estimate risks reliably. Benzo[a]pyrene is the only PAH for which a suitable database is available, allowing quantitative risk assessment. The EPA unit risk factor of lung cancer for BaP is $1x10^{-7}$ per μg/m^3 (US EPA, 1991). Limitations in the use of benzo[a]pyrene as an indicator of PAH toxicity in air pollution are that some PAH is bound to particulates, and that some of the gaseous components are not included. WHO (1987) estimated a URF of $8.7x10^{-8}$ per μg/m^3; *i.e.* almost identical to that used by US EPA.

Morbidity and heavy metals

93. Searle (2005) presents a review of epidemiological findings on the links between heavy metals and health generally. Searle gives an appraisal of the degree to which risk functions linking exposure to health endpoints are robust. Table 12 shows the linkages between heavy metals and health endpoints that are judged to have been shown to have strong evidence. These include cancers as well as hypertension (Hu *et al.*, 2007).

94. Significant non-cancer morbidity endpoints include impacts on the IQ of children from lead and mercury and the increased risk of osteoporosis from exposure to cadmium.

Table 12: Main epidemiological links between heavy metals and health endpoints

Metal	Health endpoint (relative severity of impact)	Route of exposure	Risk function
Arsenic	Skin cancer (1.5)	Ingestion/ (Inhalation?)	Increase in risk/(μg/day) = 0.002% (Risk/(ugm^{-3}) = 0.04%
	Lung cancer (1)	Inhalation	Increase in risk/(μgm^{-3}) = $1.5x10^{-3}$
	Bladder cancer (1.5)	Ingestion/ Inhalation	Increase in risk/(μg/day) = 0.01% (Increase in risk/(μgm^{-3}) = 0.0004%)

	Cardiovascular mortality (1)	Ingestion	Increase in risk/(µg/day) = 0.3%**
		Inhalation	Increase in risk/(μgm^{-3}) = 2%***
	Still birth/adverse pregnancy outcome (2)	Ingestion/ Inhalation	Increase in risk/(µg/d (Increase in risk/(μgm^{-3}) = 20%)
Cadmium	Osteoporosis (2)	Ingestion/ Inhalation	Risk/(ug/day) = 0.8% (Risk/(μgm^{-3}) = 16%
	Renal dysfunction (2.5)	Ingestion/ Inhalation	Risk/(µg/day) = 0.04% (Risk/(μgm^{-3}) = 0.8%
Chromium VI*	Lung cancer (1)	Inhalation	Increase in risk/(μgm^{-3}) = $4x10^{-3}$
Lead	Children's IQ	Ingestion	IQ points/ (µg/day in food) = 0.32 IQ points/ (µg/day in water) = 0.24 (IQ points/(µg/L) = 0.48)
		Inhalation	IQ points/(μgm^{-3}) = 0.1
	Anaemia (2.5)	Ingestion	Risk/ (µg/day in food) = 0.0048% Risk/ (µg/day in water) = 0.0096% (Risk/(µg/L) = 0.02%
		Inhalation	Risk/(μgm^{-3}) = 0.13%
	Cardiovascular illness	Ingestion/ Inhalation	Not currently established[3]
Mercury	Children's IQ	Ingested methyl mercury	IQ points/(µg/day maternal intake) = 0.149 (maximum estimate 2.8 IQ points/(µg/day)****
	CNS effects in adults – ataxia (2)	Ingested methyl mercury	Risk/(µg/day) = 0.13%
		Inhaled Hg vapour	Risk/(μgm^{-3}) = 0.015%
	Renal dysfunction – preclinical effects (3)	Inhaled Hg vapour	Risk/(μgm^{-3}) = 0.2%
Nickel	Lung cancer (1)***	Inhalation	Increase in risk/(μgm^{-3}) = $3.8x10^{-4}$

*Chromium VI accounts for a relatively small proportion of total airborne chromium
Alternative exposure-response function available – absolute risk/(µg/day) = 0.0025% - implying a fairly similar level of risk * implied 3-fold difference in risk between ingestion and inhalation seems unlikely ****Substantial uncertainties in source information
Source: Searle (2005)

95. An increasingly important area of interest is in the thresholds that may exist in the effects that pollutants have on health. Searle (2006) investigates the potential threshold effects and then identifies changes to the risk functions. Searle notes that this is not exhaustive and that many studies do not examine issues of thresholds. This is an area that needs much further research.

[3] See www.epa.gov/iris/subst/0277.htm.

Table 13: Exposure response functions and thresholds

Metal	Health endpoint (relative severity of impact)	Route of exposure	Threshold*	Risk function
Cadmium	Renal dysfunction (2.5)	Ingestion/ Inhalation	8 µg/day 0.4 μgm^{-3}	Risk/(µg/day) = 0.04% (Risk/(μgm^{-3}) = 0.8%
Lead	Anaemia (2.5)	Ingestion	Food – 300 µg/day Air – 30 μgm^{-3}	Risk/ (µg/day in food) = 1.6% Risk/ (µg/day in water) = 1.3% (Risk/(µg/L) =) = 0.7%
		Inhalation		Risk/(μgm^{-3}) = 1.33%
Mercury	Children's IQ	Ingested methyl mercury	2.8 µg/day	IQ points/(µg/day) = 0.93
	CNS effects in adults – ataxia (2)	Ingested methyl mercury	50 µg/day	Risk/(µg/day) = 0.6%
		Inhaled Hg vapour	20 μgm^{-3}	Risk/(μgm^{-3}) = 1.6%
	Renal dysfunction – preclinical effects (3)	Inhaled Hg vapour	15 μgm^{-3}	Risk/(μgm^{-3}) = 1.6%

Source: Searle (2005). *Threshold additive across all routes of exposure.

Pesticides

96. Pesticides are associated with both acute and chronic health effects. In Europe and elsewhere, agriculture workers, bystanders and consumers have the potential for acute exposures to pesticides through multiple pathways. These include exposures to workers from handling and applying pesticides, to agricultural workers re-entering treated fields, to bystanders who may live or work adjacent to agriculture fields. Additionally, consumers have the potential for acute exposure through the consumption of treated produce, *e.g.* fresh fruits and vegetables or through drinking water.

97. The impacts of chronic exposure include endocrine disruption, cancer, liver lesions and impacts on the nervous system (Hansen *et al*., 2010). However, dose-response relationships specific for active substances or pesticides classes are incomplete. Hansen *et al*. identify some studies that have examined the impact of exposure to unspecified pesticides, as shown in Table 14, though it is important to note that these health effects are indicative rather than comprehensive. They also note the potential for confounding factors and combination effects, for example the impact of predisposing factors such as smoking and gender for the risks of Parkinson's disease.

Table 14: Impact of pesticides exposure on health

Condition	Impact of exposure	Study
Parkinson's Disease	Combined odds ratio of 1.42 (95% CI 1.05-1.91) US studies: 1.72 (95% CI 1.20-2.46).	Priyadarshi *et al.* (2001)
Pregnancy-induced hypertension	Adjusted odds ratio of 1.27 (95% CI: 1.02-1.60) for residential exposure to pesticides, 1.60 (95% CI: 1.05-2.45) for agricultural exposures	Saldana *et al.* (2009)
Preeclampsia	Adjusted odds ratio of 1.32 (95%CI: 1.02-1.70) for residential exposures to pesticides, 2.07 (95%CI: 1.34-3.21) for agricultural exposures	Saldana *et al.* (2009)

Source: Based on Hansen *et al.* (2010).

Pesticide and children's health

98. Zahm and Ward (1998) reviewed the epidemiological studies linking parental and child exposure to pesticides with several types of cancer, such as leukaemia, neuroblastoma and cancer of the brain and colorectal. Most of the results reviewed by Zahm and Ward regard parental exposure to pesticides through agricultural use or children's exposure in gardens or dealing with animals. The authors summarised the

results of cross-sectional, case-control and cohort studies to conclude that although these studies have been limited by non-specific pesticide exposure information, small numbers of exposed subjects, and the potential for case-response bias, many of the reported increased risks are of greater magnitude than those observed in studies of pesticide-exposed adults. It suggests that children may be particularly sensitive to the carcinogenic effects of pesticides (in that they may have a greater susceptibility). These findings have been confirmed by subsequent studies (*e.g.* Infante-Rivard and Weichenthal (2007), whilst Eskenazi *et al.* (1999) identified a range of other potential health effects including neurodevelopmental and respiratory impacts. Recently, studies have observed that the consumption of organic fruits, vegetables and juice can significantly help to reduce children's exposure to (organophosphorus) pesticides (*e.g.* Curl *et al.*, 2003).

99. Table 15 below summarises the principal epidemiological evidence linking chemicals to health impacts.

Table 15: Exposure and dose response relationships: complex pollutants.

Pollutant	Exposure route	Exposure time [years]	Population group	Effect
As	Inhalation	70	All	Skin cancer
As	Inhalation	70	All	Lung cancer
As	Ingestion (food)	70	All	Fatal cancer
As	Ingestion (water)	70	All	Fatal cancer
As	Inhalation	70	All	Bladder cancer
As	Inhalation	35	All	Cardiovascular mortality
As	Inhalation	1	All	Still birth
Cd	Inhalation	70	All	Lung cancer
Cd	Inhalation	35	All	Osteoporosis
Cd	Inhalation	35	All	Renal dysfunction
Cd	Ingestion (food)	35	All	Osteoporosis
Cd	Ingestion (water)	35	All	Renal dysfunction
CrVI	Inhalation	70	All	Lung cancer
Ni	Inhalation	70	All	Lung cancer
Pb	Inhalation	5	Minors	IQ points loss in children
Pb	Ingestion (food)	1	Age (0,1)	IQ points loss in children
Pb	Ingestion (water)	1	Age (0,1)	IQ points loss in children
Pb	Inhalation	1	All	Anaemia
Pb	Ingestion (food)	1	All	Anaemia
Pb	Ingestion (water)	1	All	Anaemia
MeHg	Ingestion (food)	1	Minor	IQ points loss in children
Hg	Inhalation	35	All	CHS effects in adults – ataxia
Hg	Inhalation	35	All	Renal dysfunction - preclinical
PCB	Inhalation	70	All	Cancer
PCBs	Ingestion (food)	70	All	Fatal cancer
PCBs	Ingestion (water)	70	All	Fatal cancer
PCDDs	Ingestion (food)	70	All	Fatal cancer

2.2 Monetary values for health impacts related to chemicals

100. The following sub-sections summarise the available empirical evidence relating to the monetary valuation of health end-points associated with chemicals, as identified above. A previous study by Hunt (2008) provides the majority of the material for this section.

Lung cancer

Medical treatment costs

101. A review by Scasny *et al*. (2008) of the medical treatment costs associated with lung cancer identified twelve studies, ten of which were undertaken in Europe. The seven studies undertaken in Northern Europe derived a range of values for the medical costs attributable to a case of lung cancer of USD 11 200 - 27 800. One study in Southern Europe (Spain) produced a central value of USD 4 600, whilst the most recent study by Scasny *et al.* in the Czech Republic reached a central value of USD 11 000. In Canada, a study by Demeter *et al.* (2007) identified median non-small cell lung cancer and small cell lung cancer case costs to be USD 11 000 and USD 15 500, respectively.

102. 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.

Loss of productivity

103. Three studies – Weissflog *et al.* (2001), Serup-Hansen *et al.* (2003) and Scasny *et al.* (2008) – include the costs of productivity loss alongside medical costs. However, again, the assumptions adopted by the studies to derive the productivity loss costs mitigate against an easy comparison. The Weissflog *et al*. (2001) study produces a value of USD 273 000 whilst Serup-Hansen *et al.* (2003) produce a value of USD 27 000. The range of estimates produced by Scasny *et al.* (2008) of USD 59 000 to USD 200 000 are therefore in the middle of this range; we suggest, then, that a value of USD 70 000 is a reasonable central value, with the two extreme values from the other studies defining the range.

Welfare loss

104. Five studies have derived willingness-to-pay values for the intangible pain and suffering associated with lung cancer specifically. Three of the studies have been undertaken in Europe; two are from Taiwan. It is interesting to note that the research forming the basis of the three European studies was carried out at least 15 years ago. Their findings are summarised in Table 16.

105. How do we evaluate these studies relative to each other? Peer review of the study is one criteria; unfortunately for us, the two studies that are relatively recent and European – two primary criteria since, ideally, we would like to minimise spatial and temporal transfer – are those that appear not to have been peer-reviewed. These two studies have similar results, though they are derived in different ways; for example, Jeanrenaud and Priez use a private good payment vehicle (a vaccine), whilst Aimola uses a public good vehicle (a preventative health program). Three of the CV studies find scope insensitivity across different sizes of risks, whilst the other – Jeanrenaud and Priez (1999) – does not test for this. The latter study does, however, have a significant advantage in its relatively high sample size of 757 respondents. Unlike the Aimola and Hammitt & Liu studies, this study did not ask respondents to make trade-offs between other forms of cancer; we may see this as a merit in limiting their cognitive burden.

Table 16: Studies that estimate the WTP to avoid lung cancer

Study ref. *(data year if known)*	Peer reviewed	Good valued	Location	Valuation method	Sample size	Results (mean USD 2010)
Jeanrenaud and Priez (1999)	No	95% risk reduction of contracting lung cancer	Switzerland	CV (Payment card) Private good	757	VSC 0.37m – 0.43m
Aimola (1998)	No?	50 % risk reduction of death from cancer	Sicily	CV (OE & Payment card versions) Public good	89	VSL 0.44m
Åkerman, Johnson and Bergman (1991)	Yes	50 % risk reduction of lung cancer	Sollentuna, Sweden	Avertive behaviour	317	VSL 0.26m (40-year old, 3% d.r.)
Jan *et al.* (2005)	Yes	50 % risk reduction of lung cancer	Taiwan	CV Private good	328	VSC 0.015 – 2.5m
Hammitt & Liu (2004)	Yes	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)
Cameron *et al.* (2008)		1/1000,000	US	CE Private & Public goods	1619	VSC 1m

VSL = Vale of Statistical Life; VSC = Value of Statistical Case of illness.

Skin cancer

Medical treatment costs

106. Serup-Hansen *et al.* (2003) has estimated the direct and indirect costs of skin cancer (non-melanoma type, ICD code C44). They assume that all patients are treated within one year and that non-melanoma is non-fatal. Costs of hospital services are based on the prevalence approach, while costs for primary care services are based on incidence approach. Some 70% of patients are treated in primary care sector only, whilst 30% are additionally referred to treatment in hospital. Average primary care sector costs are USD 125 per case whilst costs for the 30% that require combined primary and secondary care sector treatment are USD 1 163. These costs can be distributed over a 4 year span of treatment as shown in Table 17.

Table 17: Distribution of skin cancer medical treatment cost over 4 years

Year	US $ PPP
Year1	1,192
Year2	43
Year3	26
Year4	26
Total	1,288

107. Dickie and Gerking (1996) also report estimates made by the US EPA (1987) of medical treatment costs associated with non-melanoma skin cancer, of a range of USD 5 300 to USD 9 300, significantly higher than those reported by Serup-Hansen *et al*.

Loss of productivity

108. The same study by Serup-Hansen *et al.* used expert judgement to estimate that, on average, inpatient hospital services took 4.5 days, followed by 14 days of incapacity from the work-place. One-third of a day is assumed to be lost for each outpatient hospital visit. Based on a productivity loss of USD 75 per day, average production loss was estimated to be USD 701 per skin cancer patient. It was assumed that these costs were levied during the first year following diagnosis.

Welfare costs

109. Table 18 summarises the principal studies that have produced empirical estimates of the willingness-to-pay to avoid skin cancer. The purpose of the Bateman *et al.* (2005) and Bateman and Brouwer (2006) studies was to explore the influence of exogenous risk factors in determining WTP, rather than placing any emphasis on the absolute values of the WTP.

110. Two early studies undertaken in the US – Dickie, Gerking and Agee (1991) and Dickie and Gerking (1996) – estimated marginal willingness-to pay-for reduction of skin cancer risk on a sample of 291 respondents from Wyoming and California, using a private good – sunscreen. The first study reports that the marginal willingness-to-pay for a 1% reduction in skin cancer risk lies in the USD 3.3 – USD 6.8 range for each of six age groups applying a 5% discount rate, USD 2.5 – USD 3 if applying a 10% discount rate and USD 2.0 – USD 2.5 with a 15% discount rate. These values roughly equate to a range of USD 200 to 680 per case of skin cancer avoided if we assume that the WTP for 1% is linear and proportional for all subsequent risk reductions prior to entire risk elimination.

111. In the second study, results from a WTP regression were used to compute option price estimates to reduce the risk of skin cancer for low, medium and high income households with different levels of initial perceived risk of getting skin cancer. Option prices for a five percentage point reduction in risk ranged from USD 51 to USD 76 for low income households, from USD 52 to USD 77 for medium income households and from USD 60 to USD 84 for high income households. These values are equated to a value per case of skin cancer avoided of between USD 950 and USD 1 600.

112. A further study by Dickie and Gerking (2003) surveyed 610 parents in Hattiesburg, Mississippi, US, in order to elicit relative weights between parent and child WTP and fatal and non-fatal cancers. The weightings were found to be 1:2.3 and 20:1, respectively.

113. Apart from the different methodological foci of these studies, it is also the case that different risk reductions are being valued. Indeed, the only study that explicitly states a WTP for a case avoided is Dickie and Gerking (1996). In this study, the authors also recognise that the values may be considered in potential policy analysis. The temporal and spatial transfer issues that arise in suggesting this range of values in other countries are likely to be significant.

Table 18: Studies that estimate the WTP to avoid skin cancer

Study ref. *(data year if known)*	Peer reviewed	Good valued	Location	Valuation method	Sample size	Results (mean USD, 2010)
Murdoch & Thayer 1990)	Yes	Skin cancer case	US	Avertive behaviour	Not applicable	0.046 million
Dickie, Gerking & Agee (1991)	Yes	1% redn. of lifetime risk of skin cancer	US	CV	291	200-680 per case of skin cancer
Dickie & Gerking (1996)	Yes	5% redn. of lifetime risk of skin cancer	US	CV	291	950-1 600 per case of skin cancer
Dickie & Gerking (2003)	No	Redn. of lifetime risk of skin cancer (parent & child)	US	CV Payment for sun-cream	610	Child-parent ratio = 2.33:1; mortality-morbidity ratio = 20:1
Kahneman & Ritov (1994)	Yes	Skin cancer	US	CV	1 441	15
Bateman & Brouwer (2006)	Yes	Skin cancer	US	CV	251	34-134 (OE) 249-836 (DC)
Bateman *et al.* (2005)	Yes	100% risk reduction Skin cancer	Eng, Scot, Portugal, NZ	CV	739	7-16 for sun cream product; 26-229 for international fund to reduce LDCs CFC pollution.

Leukaemia

Medical treatment costs

114. A number of studies report medical treatment costs associated with leukaemia. They include: Rahiala *et al.* (2000), Barr *et al.* (1996), Tennvall and Nilsson (1994) and Welch *et al.* (1989). These studies made empirical estimates of alternative treatments. The treatment costs appear to vary according to the type of treatment and the age group – adults or children – treated. Redaelli *et al*. identify the differences in key assumptions associated with BMT, including the duration of follow-up treatment; cost of blood; cost of drugs; laboratory costs, and; medical staff costs. Additionally, cost differences are likely to result from the location and time of the research. It is possible, however, to establish a cost range on the basis of these estimates between USD 60 000 and US 250 000. A mid-point value of US 150 000 may then be used as a central estimate.

Loss of productivity

115. One study – Tennvall and Nilsson (1994), reported in Redaelli *et al.* (2004) – estimates the productivity losses associated with leukaemia, in Sweden. Though details of how costs were derived are missing, their central estimate was USD 8 000 per case.

Welfare costs

116. We identified two studies that make estimates of the WTP associated with leukaemia: Kahneman and Ritov (1994) and Aimola (1998). They are referred to previously in the discussion of lung cancer. The study by Kahneman and Ritov may not be seen to produce WTP results valid for use in health impact assessments or CBA since it has a purely methodological focus. The study by Aimola (1998) used 1994 data obtained from 89 personal interviews conducted in Lentini, Sicily, to estimate the monetary value of

changes in the risk of death from cancer. A 50% reduction of the risk of death from four specific types of cancer was considered: leukaemia, lung cancer, uterus and prostate cancer. Willingness-to-pay was elicited using contingent valuation method: open ended format and payment card approach. The value of a statistical life for leukaemia ranges from USD 1.3 million (median) to USD 4.5 million (mean).

117. The evidence summarised in this, and proceeding, sub-sections on the monetary valuation of various forms of cancer aims to reflect current state of knowledge in this area. The following questions may be asked when considering the potential use of this body of evidence in health impact assessment and subsequent policy appraisal:

1. Is there evidence for a "cancer premium" to be added to the VSL or VOLY estimates currently used in appraisal?
2. Is there evidence that any such premium should be differentiated on the basis of type of cancer?

118. Subsidiary issues include: the validity of empirical evidence based on risk-risk values relative to those derived using risk-dollar trade-offs; the treatment of morbidity in cancer valuation; the treatment of latency in cancer valuation, and; the scope for spatial and temporal transfer of existing values.

119. In order to identify a cancer premium, it is necessary to compare the WTP to avoid a cancer risk with the WTP to avoid a risk equivalent in every other way to the cancer risk, with the exception of specific cancer characteristics. In other words, a cancer WTP has to be compared with a benchmark WTP. The comparison may be made either within a study (intra-study), with the advantage of utilising a common methodology and a similar or identical sample, or between studies (inter-study).

120. Our review shows that a small number of intra-study comparisons have been undertaken. These studies do not, however, lead to a consensus. For example, whilst the Magat *et al.* (1996) study finds no evidence of a cancer premium, the recent Van Houtven *et al.* (2008) study identifies a substantial risk premium. Both studies use risk-risk trade-offs between fatal cancers and a road accident fatality. Also, they were both undertaken in the US. Weaker evidence for a cancer premium is suggested by Hammitt and Liu (2004), Jones-Lee *et al.* (1985) and Savage (1993), all of whom find that a signal for such a premium exists but is not statistically significant. The Tsuge *et al.* (2005) study, like Magat *et al.* (1996), shows no sign of a cancer premium. However, as with the Van Houtven *et al.* study, Tsuge *et al.* identify a significant discounting of future impacts, suggesting that the latency characteristic of some cancer incidence is important. Moreover, the evidence from Savage suggests that the "unknown" and "dread" characteristics associated with cancers do exist in the minds of respondents and can be important in determining the WTP.

121. Identification of a cancer premium using the results of comparable studies is complicated firstly by the fact that a number of studies value the reduction of a risk of contracting cancer without separating out the WTP to avoid the ill health associated with the cancer from the risk of death. Jeanrenaud and Priez (1999) and Dickie and Gerking (1996) are examples of this, where both value a statistical case of cancer, in the context of lung cancer and skin cancer, respectively. Magat *et al.* (1996) do, however, investigate this issue explicitly; they find that the mortality component is about 60% of the total utility gain from the reduction in cancer (lymphoma) risk. In addition, the medical treatment and productivity loss costs of both the morbidity and mortality components need to be considered.

122. In fact, an inter-study comparison between those that produce cancer VSLs and those that produce non-cancer VSLs is likely to be inconclusive. The non-cancer benchmark adopted in the intra-study comparisons tends to be road-accident focussed, presumably on the basis that these risks lack cancer-specific characteristics such as dread. However, evidence of VSLs from the road accident context shows a very wide range of values, reflecting the significant uncertainties that exist in these applications of non-market valuation techniques. For example, European Commission (1995) reviews early studies and finds

ranges of values for transport accident VSLs of USD 6.1 million – USD 10 million (2010 prices) for stated preference applications and USD 1.1 – USD 5.3 million for avertive behaviour applications. By way of comparison, directly derived cancer VSLs include the range of values from USD 0.44 million to USD 6 millin in Aimola (1998), whilst Hammitt & Liu (2004) and Tsuge *et al.* (2005) find cancer VSLs of USD 1 million – USD 3 million. These results suggest, at best, a broad convergence of VSLs – cancer or non-cancer – with the more recent stated preference study results showing declining values. The wide range of VSLs derived, reflecting the underlying measurement uncertainties, do not generally permit a cancer premium to be identified on an inter-study basis, though such a study has recently found evidence for this premium (Navrud *et al*., 2011).

123. The second question – can WTP values be differentiated according to the type of cancer considered – raises a number of the same issues, particularly that of the high degree of uncertainty in the WTP values. The evidence presented with regard to specific types of cancer – lung, skin, leukaemia, lymph and liver – suggests that WTP values can be identified for these cancer types. Moreover, a number of studies, including Hammitt & Liu (2004), Aimola (1998) and Van Houtven *et al.* (2008), find significantly different values for different cancer types on an intra-study basis. However, the findings of individual studies are not obviously consistent with each other. For example, whilst Hammitt & Liu (2004) find that the WTP for lung cancer is 40% higher than liver cancer, of the four cancers that Aimola (1998) derives values for, the lung cancer WTP is by far the lowest.

124. We conclude first, that there exists empirical evidence for the valuation of cases of cancer and/or cancer fatalities for the types of cancer – lung, skin and leukaemia – that we are most interested in. Against this, however, it is clear that the evidence-base is very thin, relying on a handful of studies predominantly undertaken in the US. Moreover, it is difficult to view the evidence as robust; indeed, whilst there is some evidence for supporting the idea of a cancer VSL, it does not appear to be sufficiently strong to make a case for a cancer premium to be applied in current health impact assessments or policy appraisal.

IQ loss

125. An economic valuation of IQ change includes the following welfare components:

- *opportunity costs* in terms of lost productivity, *i.e.* decreased present value of expected lifetime earnings,
- direct *resource educational costs* related with compensatory education,
- opportunity costs of lost income during remedial compensatory education,
- *resource medical costs*, *i.e.* increased educational resources expended for a child who becomes mentally handicapped,
- *disutility* due to human development disabilities.

126. Note that these welfare components should not be seen as simply additive in determining a total welfare cost associated with a person who suffers from a loss in IQ. For instance, the incurring of medical and compensatory educational costs may – to some extent – result in a reduction of subsequent lost lifetime productivity. To the extent that this is true, it is appropriate to include these two cost components, but adjust the opportunity cost estimate downwards. Alternatively, it might be judged equivalent to subtract the medical and educational costs from the gross opportunity costs of lost productivity.

Medical treatment costs

127. Available estimates are presented in Table 19 below. It is apparent that the medical, and other, costs are not calibrated according to IQ but are – in both studies – related to the level of lead in blood. The

medical treatment that is costed is chelation therapy. The costs are assumed to be borne in the year of diagnosis and so are not discounted.

128. On the assumption that we do not know the incidence of the level of lead in blood in those children that suffer loss of IQ levels, it seems sensible to use a range of values that correspond to the lowest and highest values given in the Mathtec study: USD 428 and 4 400 per child respectively (2000 prices). In the first instance, the mid-point between these two values may be used as a central value. The mid-point value is 2 414 USD per child impacted. The values found in the US EPA study are contained within this range and so give us greater confidence that the value range is broadly applicable.

Table 19: Estimate of medical resource costs incurred by lead-poisoned children

Study	Cost element	Impact valued	Cost per child (US$, 2000)
US EPA (1985)	Medical costs	Preventing blood levels rising to 25µg/dl or above	1531
Mathtec (1987)	Medical costs; screening/ education programmes; opp. cost of parents time	Blood level > 40µg/dl; EP level > 53µg/dl	4398
		Blood level > 40µg/dl; EP level 35-53µg/dl	2196
		Blood level 21-40µg/dl; EP level 33-53µg/dl	1016
		Blood level 21-40µg/dl; EP level 0-32µg/dl	428
		Blood level 0-20µg/dl; EP level > 33µg/dl	565
		Blood level 0-20µg/dl; EP level 0-32µg/dl	428

Opportunity costs

129. Scasny *et al.* (2008) provide a review of the available evidence of the opportunity costs alongside the costs incurred by remedial education. Their starting point is the guidance provided by the US EPA (EPA, 1997) which combines the value of lifetime earnings with the estimate of percent wage loss per IQ point and subtracts the direct education and opportunity costs to result in a total net effect of IQ on earnings of USD 2 505 per IQ point (assuming the effect as estimated by Schwartz (1990)), or USD 3 410 (if a higher estimate of percentage wage loss per IQ by Salkever (1995) is assumed). US EPA (1997) then suggests using the midpoint in the analysis which is USD 2 957. These estimates are in fact sensitive on the discount rate used; the final estimate would be only USD 1 311 if a discount rate of 7% is assumed, or USD 6 879 employing a 3% discount rate. In Table 20, the economic costs using the assumptions from Salkever are presented.

Table 20: Loss in earnings and education costs from IQ loss

	Salkever, USD
i) Loss in earnings	4 090
ii) Costs of education	267
iii) Opportunity costs while in school	531
Total (i-ii-iii)	**3 292**

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

130. A literature review on IQ valuation undertaken by Spadaro and Rabl (2004) finds the following unit values:

- Lutter (2000) indicates USD 3 000 per IQ point,
- Grosse *et al.* (2002) estimate USD 14 500 per IQ point, (2000 prices)
- Muir and Zegarac (2001) estimate USD 15 000 per IQ point, (1999 prices)
- Rice and Hammitt (2005) estimate USD 16 500 per IQ point, and (2003 prices)
- Trasande *et al.* (2005) estimate USD 22 300 per IQ point (2003 prices).

131. On the basis of this review, Spadaro and Rabl (2004) and Spadaro and Rabl (2008) take US USD 18 000 per IQ point, including adjustment for purchase power parity. It should be noted that this value is derived on the basis of giving more weight to the last four studies listed since these are all based on lost earnings; Lutter (2000) does not include this component and is based solely on parents' WTP for their children not to suffer a loss in IQ.

132. In Scasny *et al*. (2008), the total economic costs are given by the sum of the opportunity costs in terms of loss of labour productivity, direct costs of remedial education and the opportunity costs related to the remedial education. They derive the total economic costs per IQ point of US USD 14 600 (pure time preference rate = 1%) or about US 6 300 (if prtr=3%), assuming the effect on subsequent productivity of schooling by Salkever, *i.e*. 0.1007 years; if the effect on subsequent productivity of schooling as derived by Schwartz was assumed, the economic costs are 90% (prtr=1%) or 84% (prtr=3%) of the costs derived from the schooling effect estimated by Salkever.

Disutility

133. Scasny *et al.* (2008) also note, though do not include in their estimates of total value above, that there are three studies from the US that value the disutility component of neuro-developmental disorders. Agee and Crocker, (1994 and 1996), estimate parents' willingness-to pay-to avoid high levels of lead in the blood of one of their children. Von Stackelberg and Hammitt (2005), using a stated preference technique, value certain developmental endpoints associated with exposure to PCB compounds via fish ingestion. These include a 6-point reduction in IQ, and a 7-month delay in reading comprehension. The studies are summarised in Table 21.

Table 21: Summary of estimates for value of human development disabilities

Author(s)	Description of Health Effect	Valuation Method	Location, Country	Year of Data	Estimated Value (USD)
Agee and Crocker (1994)	An increase in the information provided to parents corresponding to their child's body lead level	Averting behaviour	Chelsea, Somerville (US)	1985, 1978	*Parents mean WTP:* - overall=6.6 - who chose therapy=32.9 - who did not choose therapy=4.8 *Social mean WTP:* - overall=433.5 - who chose therapy=2169.9 - who did not choose therapy=317.8
Agee and Crocker (1996)	A marginal reduction and a one percent reduction in child body lead burden	Averting behaviour	Boston (US)	1985, 1978, 1977, 1976, 1975	*One part per million reduction* - overall=2.1 - who chose therapy=7.2 -who did not choose therapy=1.6 *One percent reduction* -overall=32 - who chose therapy=207.7 - who did not choose therapy=22.2

von Stackelberg and Hammitt (2005)	A small reduction in IQ and a probability of a 7-month reduction in reading comprehension	Contingent valuation - dichotomous choice	US	2005	Reduction in IQ=102.8 7-month reduction in reading comprehension=120.4

Source: EVRI database.

2.3 Summary of recent cost-benefit studies conducted on national chemical management policies

134. This section presents and overview of recent cost-benefit analysis studies relating to chemical management. The boundaries of this analysis are to evaluate policies that directly address chemicals management and not disposal (*i.e.* consideration is not made of the waste sector, for which a number of other studies exist, *e.g.* COWI, 2000, or on restoration of brownfields – *e.g.* Guerriero and Cairns, 2009).

135. IMV (2007) discusses a range of issues in the application of cost-benefit methods to the REACH proposals. It highlights the lack of detail in the presentation of analysis of CBA results, including presentation of the source of values used and a lack of quantitative assessment of uncertainty.

136. An interesting case study of the application of CBA in the context of chemical regulation is given by Burnett and Hahn (2001). They examine the regulation of arsenic content in water proposed by the EPA and find that the cost exceeds the benefits. They argue that more account needs to be made of non-quantifiable factors and that the indirect impacts of regulation in terms of diverting money away from health care could be taken into account. If the latter is taken into account, Burnett and Hahn argue, then the net benefits in terms of lives saved may be negative. Various appropriate values for mortality valuation are discussed and applied to show the sensitivity of the results to these values.

137. Building on the above study and the earlier US EPA study, Sunstein (2001) further examines the potential costs and benefits of the regulation on arsenic content in drinking water. This study highlights some of the complexities in evaluating the costs and benefits of health endpoints where there is considerable uncertainty. This presents revised cost and benefit estimates, based on consideration of uncertainty in the dose-response functions and in the valuation estimates – as well as additional concerns on the appropriate discount rates to be applied – noting that a 7% discount rate is likely too high for social benefits. Sunstein concludes that between 0 and 112 lives would be saved – with significant implications for the assessment of appropriate policy in the CBA context. It is true to say there is considerable uncertainty – but Sunstein gives this perhaps too much weight and is rather self-contradictory in suggesting that the "best point estimate" of the health benefit is none because of this inherent uncertainty. Sunstein goes on to note that considering life years saved and the latency issues in the health impacts, the benefits are unlikely to outweigh the USD 210 million cost of the project.

138. Lutter *et al.* (2001) investigate the costs and benefits of reducing mercury emissions from power plants in the US case. They do not explicitly value or quantify the health impacts – identifying neurological and IQ impacts of mercury and suggesting that these would likely be lower than the USD 1.1 billion to USD 1.7 billion annual costs of reducing emissions.

139. US EPA (2001) presents an economic assessment of the impacts of changing regulation on the wastewater relating to the paint industry. No population risks were estimated in terms of health – and so there was no estimation of the health benefits of the proposed regulation or of the monetary values.

140. Entec (2001) evaluated the costs and benefits of compliance with heavy metal limit values for the EU-15. The costs were found to significantly outweigh the benefits. This study used values from DG Environment for the Value of a Prevented Fatality, with adjustment for latency of cancer effects, a cancer premium of 50% and age. They arrived at a central value of USD 1.8 million (range USD 0.9 million to

USD 4.4 million), with an assumption that all cancers lead to fatal outcomes after a period of time. The age adjustment is not well explained and seems questionable in the context. The study does not consider ancillary health impacts of the policy – which may lead to a significant underestimation of the true health benefit. Crops and ecosystems are mentioned, but not valued.

141. The impact of REACH regulation was estimated by DHI (2005). Drawing on benefit transfer of values of USD 1 million for a fatal cancer and USD 400 000 for a non-fatal cancer (Eftec, 2004) and a willingness-to-pay study for drinking water (WRc, 1999) and applying damage functions they arrive at the damage costs associated with different chemicals as shown in Table 22. The study also shows the benefits of REACH in terms of reductions in the costs of disposal of dredged sediment (with a total benefit of USD 241 million to USD 1 450 million over 25 years).

Table 22: Costs associated with 4 case substances

	1,2,4-TCB	NP(E)	PER	PCB
Costs (mill €/year)	98-582	229-1829	0.3-0.8	78-583

Source: DHI, 2005

142. EC (2008) estimated the costs and benefits of implementing new chemical regulations on the production of toys, as well as other regulations to prevent choking. Different assumptions as to the value of a Disability Adjusted Life Year (DALY) and levels of ingestion were used to assess the benefits of the new regulation. The value of a DALY was taken to be between USD 45 000 and USD 90 000. For a period covering 2008 to 2051, a cost-benefit approach is applied. Significant benefits of USD 12.4 billion were identified for a risk-based approach to the setting of regulation, with incremental benefits of alternative policies of USD 68 million to USD 340 million, depending on the specific nature of the policy intervention. These estimates are all presented with a significant degree of uncertainty – here the mid values are presented, but the incremental benefit of the risk-based approach range from USD 1.2 billion to USD 50.9 billion, depending on the assumptions used. The overall results of the cost-benefit analysis are not clearly presented – though for the risk-based approach to the policy, the NPV of net benefits is estimated at USD 12.5 billion, which does not seem consistent with the benefit estimate reported above and estimated NPV costs of USD 5 billion. Costs for more stringent approaches are presented at USD 13.4 billion and USD 13.7 billion. These are suggested to outweigh the benefits – but because of lack of reliable information as to the scale of these costs and underestimation of the health and other benefits, the most stringent policy was proposed. This highlights a major issue in the use of cost-benefit analysis in the presence of asymmetric information on the costs of implementation of policies combined with uncertainties in the health estimation procedures.

143. The potential costs and benefits of alternative policies to restrict the marketing and use of cadmium were investigated by RPA (2010). This study examined the use of cadmium in brazing alloys, in jewellery and in PVC waste. Health benefits were quantified and monetised. For brazing alloys, health impacts included lung cancer and occupational emphysema for professionals as well as hobbyists. Incidents of lung cancer are valued using benefit transfer with a USD 1.2 million lower bound and USD 1.8 million upper bound, whilst an additional case of emphysema is valued based on UK estimates of productivity loss and treatment cost of between USD 1 100 to USD 1 600 per case. This would seem to suggest that the emphysema is considered to be an episodic cost of the disease (equivalent to an asthma attack), which would seem to not be the best way to value an additional case. RPA note the latency effects in emphysema, but do not discuss how they treat this in the analysis.

144. The overall results show that in terms of cost-benefit analysis, the evidence for the measures is mixed. For brazing alloys, the overall findings are that the present value costs of USD 7.8 to USD 147

million over 20 years significantly exceed the health benefits for hobbyists of USD 0.7 to USD 2.2 million, though this does not account for the impacts of short-time exposure to high concentrations. The benefits for professional users of USD 98 to USD 473 million are considered to far exceed the costs to industry. In the case of restrictions on the use of cadmium in jewellery, health benefits of USD 3.67-7.22 million are estimated. These are considered "modest" compared to the costs of the proposals, though not all benefits could be quantified. Analysis of PVC waste policy suggests that there would be environmental and health benefits in relaxing legislation on the limit on cadmium in PVC that can be recycled, due to the impacts of incineration and landfilling and also producing new PVC.

Table 23: Summary of CBA in Chemicals

Author(s) / Date /Place	Pollutant	Policy input/Purpose	Health Measurement
Entec UK Ltd – 2001 Europe	Arsenic, Cadmium, Nickel	Compliance with Heavy Metal Limit Values for EU15	Fatal Cancers, non carcinogens considered negligible
ERM Economics – 1996 Europe	Lead; SO_2	CBA for Integrated Pollution Control – Hypothetical industrial plant	IQ; Blood Pressure
EC (2008) Europe	Chemicals in toys	Impact assessment/CBA of policy to chemicals in toys	DALY approach
DHI (2005) Europe	REACH chemicals	Valuing benefit of REACH	Fatal and non fatal cancers
RPA (2010) Europe	Cadmium	Impact assessment of potential update to restrictions on marketing and use of cadmium	Mortality
Sunstein (2001) US	Arsenic	Assessment of costs and benefits of USEPA regulation on drinking water	Mortality
Burnett and Hahn (2001) US	Arsenic	Assessment of costs and benefits of USEPA regulation on drinking water	Mortality
Lutter *et al.* (2001) US	Mercury	CBA of cutting emissions of mercury from power plants	No formal valuation
US EPA (2001)	Paints	Cost assessment of impact of regulation on wastewaters	Not quantified

2.4 Summary and Conclusions

145. The impact of chemicals on health has been the focus of significant research. Scientific evidence of the linkage between exposure and health impact usually consists of animal studies and some epidemiological studies of workers exposed to high concentrations, which has implications for the application of results to the analysis of policies affecting exposures of the general public to toxic chemicals.

146. Cost-benefit analysis of the impacts of policies that have significant health implications in relation to chemical is made difficult by the relative lack of detail on the health linkages (*e.g.* in terms of dose-response functions) and the impact of threshold levels, though the quality of this data is improving rapidly.

147. Fairly robust epidemiological links exist for a range of exposures to heavy metals in particular and health endpoints. These include:

- Arsenic exposure and skin, lung and bladder cancers, cardiovascular mortality and still births;
- Cadmium exposure and osteoporosis and renal dysfunction;
- Chromium exposures and lung cancer;
- Lead exposure and impacts on IQ in children and anaemia;
- Mercury exposure and impacts on IQ in children, effects on the central nervous system and renal dysfunction; and

- Nickel exposure on lung cancer. (Searle, 2005)

148. Thresholds have been identified in the relationship between:

- Cadmium and renal dysfunction;
- Lead and anaemia;
- Mercury and IQ in children, impacts on the central nervous system in adults and renal dysfunction.

149. In addition, the impacts of mixed chemicals may be significantly different from the effects of individual chemicals considered in isolation.

150. In terms of valuation, some forms of cancer have been studied more than others – and there is considerable variation shown in terms of the values given different contexts and cancer types. A summary of the main welfare costs associated with chemical related health effects is given below. It can be seen that leukaemia is generally considered to have a significantly higher welfare impact than lung cancer, with skin cancer having the lowest impact in general. Neuro-developmental disorders can be valued at approximately USD 300 000 per case. It should be noted that care should be taken in the use of values in the table below for policy evaluation – as specific cancer impacts of chemical releases may have acute or latent impacts. The issue of the valuation of cancer cases in children is also controversial. Finally, it should be noted that these estimates are for a limited number of health end-points only; as identified above, there are a number of other non-cancer effects arising from chemicals pollution.

Table 24: Summary of welfare costs associated with chemical-related health impacts (USD 2010)

	Medical treatment costs	Productivity loss costs	Disutility (value of a case)	Total WTP
Cancer (Lung)	11 000 (4 600 – 27 800)	70 000 (27 000 – 273 000)	400 000 (15 000 – 2 500 000)	481 000 (46 600 – 2 800 800)
Skin cancer	1 300 (125 – 9 300)	7 000	1 000 (200 – 1 600)	9 300 (7 325 – 17 900)
Leukaemia	150 000 (60 000 – 250 000)	8 000	2 500 000 (1.3 million – 4.5 million)	2.658 million (1.368 million – 4.758 million)
Neuro-devt. disorders	2 500 (450 – 5 000)	7 500 (2 500 – 18 000)	-	302 500 (100 000 – 725 000)

151. The review of studies that have considered health as part of detailed cost-benefit analysis shows a number of major issues. These include:

- The treatment of ancillary impacts of chemical regulations is limited. Actions to mitigate chemical releases are likely to have impacts on other pollutants and these should be considered where possible.
- Values applied for health impacts vary – even in studies funded by the European Commission where more of a standardised "unit value" for a VPF has been applied, there is marked variation in the treatment of latency, the cancer premium and the treatment of age. There is sometimes inconsistency in the values applied for health endpoints and the nature of the endpoint.
- The presentation of quantification methods and studies used to derive values is rather mixed. Variations in assumptions such as discounting affect the comparability of results.
- The application of the valuation of morbidity endpoints in the analysis of chemicals policy is limited.
- Sensitivity analysis has been based around the use of upper and lower values, and "best guess" values. This may be because of the time frame involved in conducting these analyses, which are

often driven by regulatory timetables that are quite short. Advanced quantitative analysis of uncertainties using *e.g.* Monte Carlo methods is seldom conducted.

- Unquantified health impacts are sometimes used to justify policies with significant costs – *e.g.* in EC (2008) cost-benefit analysis is used to justify increased regulations on the production of toys, with the most stringent regulation being proposed despite a cost of over USD 13 billion. Given the costs, further research on the unquantified health impacts – even using simple expert judgement or Delphi methods to do some quantification – may have been justified.

152. The impact of chemical exposure on health is an area that needs significant further research. Little is known of the impacts of exposure to multiple chemicals on health. In terms of valuation, certain endpoints have attracted more attention than others – there is need for further studies to value morbidity endpoints in particular. Finally, there is the need for improved consistency in the application of cost-benefit analysis methods in the context of policies to address exposure to chemicals – to enable transferability of results between contexts and to ensure state-of-the-art methods are used in policy evaluation.

153. There are a number of concrete suggestions that can be made at this point. First, the possibility of promoting a database that provides a set of common unit values for use in chemical policy appraisal should be explored. Such a database could build on previous initiatives such as the BeTa Methodex model,[4] developed as part of the EC Methodex research project, and which facilitated the calculation of cost per unit of pollutant emission. It would also work to convert the outputs of the EVRI valuation database[5] to an application-ready form.

154. To complement this, it is suggested that the recent guidance on value transfer prepared for OECD in the context of mortality risk valuation should be generalised to cover non-fatal health impacts, and made available – with a number of worked examples – as a common approach to be adopted.

155. Further, it is suggested that this template could be developed to aid the transfer of exposure-response functions. Any such guidance should explicitly illustrate how uncertainties in the physical and monetary estimation process should be represented consistently in health impact assessments and cost-benefit analysis applications.

156. Whilst these suggestions are made in the context of analysis of chemicals, they remain applicable to both the air and water & sanitation sections, where there remains a similar need for collation, communication and consistency in the use of data and analytical tools such as CBA.

3. Unsafe water supply and sanitation (WSS)

3.1 Health impacts associated with WSS

157. The empirical evidence relating to the quantification of the human health impacts associated with water supply and sanitation (WSS) is fairly well documented. Whilst epidemiological research is investigating the precise nature of the linkages between environmental risk factors and health impacts, (see *e.g.* McMichael *et al*., 2001), attribution between risk factors is complex. In particular, the range of possible alternative pathways by which diseases are transmitted makes attribution difficult. For example, Prüss *et al.* (2002) highlight that human and animal excreta can affect human health in the form of a number of different diseases through drinking water, sewage, indirect contact, and food through various pathways. It is, however, established that these faecal-oral diseases comprise the majority of the disease

4 Some details of this work remain at wwwb.vito.be/reach_sea_datasources/view_details.aspx?link=172. The database was created by Mike Holland of EMRC.

5 www.evri.ca.

burden resulting from WSS (Fewtrell *et al*., 2005). The vast majority of this disease burden is borne by lower income countries. Prüss-Üstün *et al*. (2008) estimate that the diseases associated with poor sanitation are particularly correlated with poverty and infancy and alone account for about 10% of the global burden of disease. Water quality is identified as being the second largest contributor to the global burden of disease (Hunter *et al.*, 2010). Hunter *et al.* also highlight that most of the excess disease burden in developing countries falls on young children – 17% of all deaths in children under 5 years are attributed to diarrhoea. In higher-income countries, attention has recently been given to the potential health impacts arising from disinfection by-products. For example, it is thought likely that the use of chlorine results in trihalomethanes (THMs) and haloacetic acids (HAAs) that may give rise to cancer, as well as reproductive and developmental effects.

158. The existence of water supply and sewage infrastructure currently ensures that the risk factors associated with these diseases are very low in most OECD countries. However, the disease risk in these countries still exists, primarily as a result of disruption to the water and sanitation infrastructure, for example, from intra-urban flooding. Such a risk is projected to increase under climate change scenarios in OECD countries, and globally (Wilby, 2007; Douglas *et al*., 2008).

159. Table 25 identifies the principal health impacts associated with WSS. It highlights that whilst diarrhoea is a disease common to both water and sanitation-based pollutants, other serious acute and chronic diseases are associated with faecal and bacterial pollutants.

Table 25: Health impacts associated with WSS

Pollutant	Source	Health Impact
Faecal contamination	Bathing waters	Gastroenteritis, acute respiratory disease, infections, diarrhoea
Bacteria (protozoa)	Drinking water	Diarrhoea, amoebic dysentery, cholera, cryptosporidiosis
Viruses	Drinking water	Diarrhoea, gastroenteritis, meningitis, non-specific febrile illnesses

3.2 Methodology used to derive monetary values for health impacts

160. The previous sub-section serves to emphasise that there are established quantitative links between pollutants linked to inadequate water supply and sanitation, and that there is potential scope in the evaluation of options to improve these services to compare the likely benefits of such options with their costs. This sub-section therefore explores the possibilities for the monetisation of these benefits and the use of monetary values in health impact assessment of these diseases, more generally. More specifically, the principal objective is to identify the potential for value transfer of the empirical data to OECD policy analysis contexts. To meet this objective, the studies are considered in the collective though the relative merits of individual studies and their findings are highlighted.

161. Table 26 identifies a number of the principal studies that have derived values relating to the health impacts of water supply and sanitation. Only studies that explicitly separate out the health impact and associated welfare costs are included in this review. Each study is identified by location, the source and type of pollution, the health impacts addressed in the valuation exercise, the method adopted for valuation and the central results. It is clear that the number of studies is limited, thereby constraining the opportunities for cross-comparison. This constraint is perhaps exacerbated by the fact that the geographical spread of the studies is high: there are five studies from North America, four from Europe, three from Central and South America and one from Asia (China).

Table 26: Summary of valuation studies relating to WSS health impacts

Authors/year	Pollutant	Source	Methodology	Measurement and place of application	Valuation (USD, 2010)
Adamowicz *et al.* (2007)	E. coli, cryptosporidium, and giardia	Drinking water	Empirical - CV	WTP to avoid a statistical case of microbial disease (diarrhoea), Canada	33,150 to 46,040
Barton & Mourato (2003)	Sewage	Effluent	Empirical - CV	Individual WTP for avoiding 1 day of illness episode from sewage in coastal bathing waters (Costa Rica and Portugal) Eyes Gastro-intestinal Cough	Portugal – 111.78 Costa Rica – 61.82 Portugal 169.45 Costa Rica – 79.67 Portugal – 91.68 Costa Rica – 44.58
Brox *et al*. (2003)	Sewage	Urban run-off	Empirical - CV	WTP for improved water quality, per household/month, Canada	2.42-4.73
Dasgupta (2004)	Microbial diseases	Drinking water	Cost of Illness	Health treatment and lost productivity costs per case per household. Delhi, India	5
Georgiou *et al*. (1998)	Sewage	Effluent	Empirical -CV	WTP for reduction in risk of illness from quality of sea bathing water, per individual per annum, England	13.49 – 20.74
Georgiou *et al.* (2000)	Sewage	Effluent	Empirical - CV	WTP for new EC standard, per household, per annum (Sea bathing water), England	71.47
Hajkowicz (2006)	Various	Urban run-off	Empirical – cost-based	Household avoidable costs per year, New Zealand	USD million, Total health costs 1.15
Hardner (1996)	Various Micro-organisms	Human and animal waste	Empirical - CV	Ecuador WTP for potable water per household per week	5.16
Jerrett (1996)	Various	Various	Empirical –cost-based	Defensive expenditures per capita per annum, Canada	530 – 1 395
Machado and Mourato, (2002)	Sewage	Sewage waste	Empirical – CV and CR	WTP to avoid a case of Gastroenteritis per person from coastal bathing, Estoril, Portugal.	57.7
Mourato *et al.* (2003)	Micro-organisms	Human/ animal waste	Empirical - CE	WTP for reducing risk of stomach upset per household, per annum, England	1.3-2.6
Ozdemiroglu *et al.* (2004)	Sewage	Sewage overflow	Empirical – CV and CE	WTP for reduction in sewage overflows per household, per annum, London, England	68.37
Soto Montes de Oca *et al.* (2003)	N/S	N/S	Empirical - CV	WTP for improvement in water quality per month, Mexico	12.9
Dwight *et al.* (2004)	NS micro-organisms	Urban run-off	Cost of Illness	Cost of : gastro-intestinal episode Acute respiratory disease Ear infection episode Eye infections episode USA	 43.16 90.58 44.67 32.23

Anderson *et al.* (2000)	Harmful algal blooms	General effluent	BT	Cost of illness per case food (fish) poisoning, USA VSL	1 650 5.68 million
Zhang (1999)	Various	Waste water	BT	Health benefits of waste water treatment, China Productivity loss per day Hospital cost per day	 3 30

CV = Contingent valuation; BT = Benefit Transfer; CE = Choice experiment; CR = Contingent ranking.

162. The majority of the studies address health impacts resulting from waste-water and sewage. The health risks considered result from coastal water bathing being contaminated by sewage effluent (*e.g.* Barton and Mourato, 2003; Geourgiou *et al.*, 1998, 2000), and those from urban wastewater overflow (*e.g.* Brox *et al*., 2003, in Canada; Ozdemiroglu et al., 2004, in London, England). There are three studies that relate to the avoidance of microbial disease through safe provision of potable water to households – Adamowicz *et al*. (2007) for Canadian households, that of Dasgupta (2004) for Delhi, India, and Hardner (1996) for village households in rural Ecuador.

163. The treatment of health differs a) to the extent that the specific health condition is identified and valued, and; b) according to the component of welfare costs that are addressed. For example, with respect to (a), studies such as Georgiou *et al*. (1998) and Brox *et al*. (2003) measure welfare changes with respect to an overall change in the risk of illness from the pollution source or receptor – coastal bathing waters and urban run-off, respectively. Other studies, however, including Machado and Mourato (2003) and Barton and Mourato (2003) identify WTP to avoid cases of specific illnesses associated with water pollution. With respect to (b), the studies divide between those, such as Machado and Mourato (2003), that estimate the WTP to avoid illness and the pain and suffering implied, and those studies that estimate the direct economic costs from lost productivity and expenditure on medical treatment. These latter cost-of-illness based studies include Dasgupta (2004), Hajkowicz (2006), Dwight *et al.* (2004) and Zhang (1999). Since the total welfare costs of a health impact are generally assumed to be the sum of the costs of illness and the WTP to avoid the pain and suffering, it is clear that these estimates are currently incomplete. However, it is also not possible simply to sum those estimates that we have identified because they have not been estimated for a common illness type.

164. The studies highlighted in Table 26 and discussed briefly above are clearly very disparate in nature, making tests of coherent validity impossible and making it difficult for us to recommend unit values for use in policy analysis. The studies' context-specificity and methodological differences exacerbate this difficulty.

165. A number of these studies have not been peer-reviewed and are taken from the grey literature. Those that have been peer-reviewed include Georgiou *et al*. (1998, 2000); Dwight, (2004); Hardner, (1996); and Brox *et al*. (2003). Whilst the fact that some studies have been peer-reviewed, and others have not, cannot be taken to be a proof of difference in the quality of the studies, it acts as one indicator of quality of the empirical values generated.

166. However, the following suggestion can be made with regard to the use of the existing empirical estimates. Whilst the preceding paragraphs suggest a number of caveats in relation to the interpretation of these estimates, their transfer and use in other policy assessments is to be encouraged, in the absence of other context-specific data. The main basis for this conclusion is in the broad convergence of the WTP estimates that exist for a common endpoint. One example is for gastroenteritis. For this illness, Barton and Mourato (2003) find WTP to avoid a day of gastroenteritis of around USD 170 in Portugal and 80 in Costa Rica. Machado and Mourato (2002) derive a value of USD 58 in Portugal. In this case, an indicative central value of USD 100 may be reasonable. It may then be possible to add to that the cost of illness estimate for gastroenteritis made by Dwight *et al*. (2004) of USD 44 in USA.

167. The main constraint we would place on this process of transfer is that whilst the unit values are expressed in common units through using US dollars purchasing power parity (PPP) equivalents, the use of cost of illness estimates should be sense-checked where transfer between regions is envisaged. This accords with the findings of Ready *et al.* (2004) who 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.

168. A further constraint would be to ensure that the unit values for specific health endpoints only should be used, in conjunction with the appropriate risk factor.

3.3 Summary of recent policy interventions

169. The range of policy interventions available to reduce pollution from water and wastewater are well established. They are summarised in Table 27. At the household level, the principal intervention is to provide access to safe supply of water. Options available to achieve this include well construction and maintenance at the local level and/or the construction of water transport and distribution networks at the municipal level, with associated water treatment facilities. Water treatment is then necessary to remove biological and chemical pollutants. In many OECD countries, this is undertaken off-site at the point of source, although it may also be required at point-of-use (*i.e.* at household level), in the instance where water is at risk of contamination during transport or storage. OECD (2010b) highlight that water treatment technologies include filtration, chlorination, flocculation, solar disinfection, ozonation, distillation, ultraviolet disinfection as well as the boiling and pasteurizing of water in non-OECD countries.

170. Options to reduce waste water pollution are defined in general terms as sanitation, i.e. the "methods for the safe and sustainable management of human excreta, including the collection, storage, treatment and disposal of faeces and urine" (OECD, 2010). These include on-site sanitation systems (such as lavatories), and network-based sanitation solutions, with or without treatment of the sewage collected. Network-based water and sanitation systems are a feature of most OECD countries (Jenkins *et al*., 2009)

171. These capital investments are complemented by hygiene promotion, *i.e.* the provision of hand-washing points, hygiene and health education and the encouragement of specific behaviours such as hand washing at critical times, keeping animals out of the kitchen, proper management of child excreta and proper storage of household drinking water (OECD, 2010).

172. Cairncross and Valdmanis (2006) confirm that the level of service of water supply is likely to have a positive effect on the avoidance of diarrheal diseases. OECD (2010b) suggests, however, that it is unclear as to whether different types of latrines are more or less effective in producing health benefits. A tentative conclusion is that simple latrines can be very effective, whilst sewage captured via sewers and released untreated in the environment can result in the spread of disease.

Table 27: Policy Interventions relevant to WSS

Stage in Value Chain	Health Benefits
Providing access to safe water and sanitation, including wastewater collection and transport	Reduced incidence of diseases, especially waterborne and water-washed diseases
Investing downstream in wastewater treatment for safe disposal and reuse	Additional health benefits, including from improved quality of recreational waters
Investing upstream in managing the supply/demand balance sustainably	Increased quality of life due to reliable water supply

3.4 Summary and comparison of cost-benefit studies of WSS policy interventions

173. Table 28 below summarises a number of recent studies that have reported cost-benefit analyses of WSS options. The studies range from assessments of water supply and waste treatment at the municipal level to those at the world regional level. The varying geographical scales reflect the fact that resource allocation is determined at these different scales. Since, for example, local authorities are often liable for investment in *e.g.* water supply at the municipal scale, the cost-benefit analysis of a municipal-based rainwater harvesting system undertaken by Tang (2009) may serve to inform such local investment decisions.

Table 28: Summary of CBAs of WSS Options with Health Benefits

Study; Country/Region *Ex Ante*/*Ex Post*	Policy/ Project details	Health Impacts coverage	Treatment of Costs	Outcomes (NPV; BCR)
Whittington *et al.* (2009)	Four water and sanitation interventions: (1) rural water supply programs providing poor rural communities in Africa with deep boreholes and public hand pumps, (2) "total sanitation" (CLTS) campaigns to halt open defecation in South Asia, (3) biosand filter, a specific point-of-use water disinfection technology for household water treatment, and (4) a large, multipurpose dam in Africa.	Cases of diarrhoea: Morbidity: USD 6 (COI) Mortality: VSL USD 30 000 (USD 10 000 – USD 50 000) from Maskery *et al.* (2008)	Includes capital and variable cost components. Varying project life-time assumptions.	BCR: Rural Water: 3.2 CLTS: 2.7 Biosand filter: 2.7 Dam: 1.8
Hutton *et al.* (2007) (Incorporates related papers) 11 WHO World regions *Ex Ante*	Five types of water supply and sanitation improvement: achieving the water MDG by 2015; achieving the combined water supply and sanitation MDG; universal basic access to water supply and sanitation; universal basic access plus water purification at point-of-use; regulated piped water supply and sewer connection	Infectious diarrhoea including cholera, salmonellosis, shigellosis, amoebiasis, and other bacterial, protozoal and viral intestinal diseases. Valuation by costs of illness plus human capital based mortality valuation. Vary by region.		All water and sanitation improvements are cost-beneficial in all developing world sub-regions. BCR 5 – 46 in developing regions. For least developed regions, BCR 5 – 12.
Tang (2009)	Rainwater harvesting system in Kerala, India	Diarrhoea USD 50/capita/annum	Construction cost of rainwater harvesting system and the maintenance costs	
Jeuland and Whittington (2009) Generic developing country context *Ex Ante*	Two water supply interventions – deep wells with public hand pumps and biosand filters. Two types of cholera immunization programs with new-generation vaccines – general community-based and targeted and school-based programs, and combinations of water supply and vaccine options.	Case of diarrhoea: Morbidity: $6 (COI) Mortality: VSL USD 30 000 (USD 10 000 – USD 50 000) from Maskery *et al.* (2008) Case of cholera (COI) USD 50 (USD15-85)	Includes capital and variable cost components of all options. Varying project life-time assumptions.	BCR using average parameter values Borehole + hand pump: 3.17 Biosand filters: 2.93 Community-based cholera vaccination: 0.9 School-based cholera vaccination: 2.64
Molinos-Senante *et al.* (2010) *Ex Post*	22 wastewater treatment plants in Valencia, Spain	Four undesirable outputs (nitrogen, phosphorus, suspended solids, and chemical oxygen demand have shadow prices attached based on literature. Health benefits not separated out.	Includes capital and variable cost components of wastewater treatment	BCRs > 1 under most scenarios of water selling/not selling

Olivieri *et al.* (2005) California, USA *Ex Ante*	Evaluation of seasonally based effluent limits for wastewater treatment facilities, including introduction of disinfected tertiary treatment during the winter.	One case of gastroenteritis from recreational activities: USD 299 and USD 1 202. Derived from valuation of salmonellosis (Mauskopf and French, 1991).	Includes capital and variable cost components of additional disinfected wastewater tertiary treatment	BCR < 1 under plausible assumptions
Godfrey *et al.* (2009) *Ex Post*	Construction of greywater treatment and reuse systems in residential schools; treated greywater use for toilet flushing and irrigating the food crops: Madhya Pradesh, India.	Cases of Diarrhoea: Morbidity: COI – local data Mortality: human capital based, as per Hutton *et al.* (2007)	Includes capital and variable cost components of greywater treatment technology.	BCR of 9

174. It is also notable that two studies – Whittington *et al.* (2009) and Jeuland and Whittington (2009) – report CBAs of investment in WSS options under generic conditions, *i.e.* without being specified at a geographical location, other than being in developing countries. These studies should therefore be seen as demonstrating both methodological issues in undertaking CBAs in this sector and in illustrating that largely favourable results (Benefit Cost Ratio (BCR) > 1) may be expected to be found when these options are considered. Thus, these studies – as well as the study by Hutton *et al.* (2007) – serve to make explicit that it is the absolute levels of financing available that should be seen as the limiting factor in determining investment in WSS options in these countries.

175. This finding contrasts with that of the two studies that report on CBA undertaken on wastewater investments in more developed countries. Whilst Molinos-Senante *et al.* (2010) generally find BCRs greater than 1 for recycling wastewater in the Spanish region of Valencia, those for further, tertiary, investment in wastewater treatment in California, USA, reported by Olivieri (2005) are described as being less than 1. These findings support the previous conclusion, made in OECD (2010b), that whilst investment in WSS in developing country contexts in likely to be very favourable using economic efficiency criteria, the conclusion is much more equivocal in developed country contexts. That paper concluded that there is a "need to conduct a systematic integrated planning of investments in WSS that combines the different components of the value chain. Investments in drinking water and sewage cannot be considered in isolation of (upstream) resource protection and (downstream) wastewater treatment. Their integration allows avoiding unnecessary costs and maximising benefits along the value chain, whilst avoiding potential "disbenefits" from inadequately timed or sequenced investments".

176. Jeuland and Whittington (2009) is noteworthy in additional respects. First, it is unique in its consideration of a number of alternative WSS options rather than a single investment, and the possibility of alternative options combined together being more or less effective. Indeed they find that implementing community-based cholera vaccination programs after borehole + hand pump or biosand filters have already been installed will rarely be justified. This is especially true when the biosand filters are already in place, because these achieve substantial cholera risk reductions on their own. On the other hand, implementing school-based cholera vaccination programs after the installation of boreholes with hand pump is more likely to be economically attractive. Also, if policy makers were to first invest in cholera vaccinations, then subsequently investing in water interventions is still likely to yield positive economic outcomes. This is because point-of-use water treatment delivers health benefits other than reduced cholera, and deep boreholes + hand pumps often yield non-health benefits, such as time savings.

177. Second, Jeuland and Whittington (2009) adopt best practice in the treatment of uncertainty in the parameterisation of the CBA. They employ probabilistic sensitivity analysis, using Monte Carlo sampling techniques to estimate a frequency distribution of benefit–cost ratios for all four interventions, given a wide variety of possible parameter combinations. Whilst some of the other studies undertake sensitivity analysis using ranges for key parameters, the use of probabilistic techniques is unique in this sector.

178. These studies are selected specifically on the basis that they explicitly quantify and monetise the health impacts of the options considered. It has previously been noted, OECD (2010b), that health impacts are typically found to be a significant, but not dominant parameter in the determination of the BCR; time savings are more important in the benefits of the majority of WSS options in developing countries. However, it is noteworthy that the coverage of health impacts in the majority of the studies is limited to consideration of diarrhoea, fatal and non-fatal. Moreover, it is clear that valuation of these endpoints is partial. For instance, all non-fatal cases of diarrhoea are valued on the basis of the costs of illness; they do not include a WTP estimate for the pain and suffering component of the welfare cost that one would expect to be considerably greater than the COI component. In the case of valuing fatal cases of diarrhoea, the

studies either use the non-WTP method based on lost lifetime earnings (*e.g*. Hutton *et al*., 2007), or use a value of statistical life derived from a single study undertaken in Bangladesh (*e.g.* Maskery *et al*., 2008, in Whittington *et al*., 2009) that is below the levels derived in the majority of studies undertaken globally (Lindhjem *et al*., 2010). For these reasons, it may be expected that the health impacts are considerably under-represented in CBAs of WSS to date.

3.5 Summary and conclusions

179. Removal of pollutants and associated health risks are a key objective of a number of actions in the water supply and sanitation sector. Principal health impacts of poor water and sanitation services include risks of gastroenteritis, diarrhoea, cholera and methaemoglobinaemia. The risks of these are very low in OECD countries, though disruption of services from intra-urban flooding may result in some cases. Under climate change, the risks of flooding may increase.

180. Studies that place monetary values on the health implications of water supply and sanitation interventions are limited. The majority of the studies address health impacts resulting from wastewater and sewage, with some relating to avoidance of microbial disease. The treatment of health differs a) to the extent that the specific health condition is identified and valued, and; b) according to the component of welfare costs that are addressed. Some studies value the welfare change of a change in a risk of illness from a particular receptor or source. Others investigate the willingness-to-pay to avoid cases of specific conditions. Few studies in this area investigate willingness-to-pay to avoid illness and the pain and suffering implied, with the majority employing cost-of-illness methods. The coverage of illnesses varies – gastroenteritis and food poisoning have been the most studied. The degree to which such results can be transferred is questionable.

181. The following suggestion can be made with regard to the use of the existing empirical estimates of willingness-to-pay. Whilst there are a number of caveats in relation to the interpretation of these estimates, their transfer and use in other policy assessments is to be encouraged, in the absence of other context-specific data. The main basis for this conclusion is in the broad convergence of the WTP estimates that exist for a common endpoint. One example is for gastroenteritis. For this illness, Barton and Mourato (2003) find WTP to avoid a day of gastroenteritis of around USD 170 in Portugal and USD 80 in Costa Rica. Machado and Mourato (2002) derive a value of USD 58 in Portugal. In this case, an indicative central value of USD 100 may be reasonable. It may then be possible to add to that the cost of illness estimate for gastroenteritis made by Dwight *et al.* (2004) of USD 44 in USA. The main constraint we would place on this process of transfer is that whilst the unit values are expressed in common units, through using US dollars purchasing power parity (PPP) equivalents, the use of cost-of-illness estimates should be sense-checked where transfer between regions is envisaged.

182. A number of recent studies that have reported cost-benefit analyses of WSS options. The studies range from assessments of water supply and waste treatment at the municipal level to those at the world regional level. The varying geographical scales reflect the fact that resource allocation is determined at these different scales. Two studies report CBAs of investment in WSS options under generic conditions, *i.e.* without being specified at a geographical location, other than being in developing countries (Whittington *et al*., 2009 and Jeuland and Whittington, 2009). These studies should therefore be seen as demonstrating both methodological issues in undertaking CBAs in this sector and in illustrating that largely favourable results (BCR > 1) may be expected to be found when these options are considered. These studies serve to make explicit that it is the absolute levels of financing available that should be seen as the limiting factor in determining investment in WSS options in developing countries

183. The same is not true for developed countries. Here, more detailed CBA is required – as some cases show BCRs of investment of less than 1 for certain interventions. This suggests more specific studies are required in the developed country context before interventions are undertaken.

184. The consideration of sequencing of interventions in cost-benefit analysis and the impact of uncertainty of the analysis differs from study to study. The implications of sequencing are that certain interventions in developing country contexts may have BCR > 1 if they are implemented before other measures, whereas if they are implemented afterwards, then they exhibit a BCR<1 (*e.g.* vaccination strategies for water-borne disease and sequencing compared to infrastructure investments). One study stands out in its use of probabilistic methods for sensitivity analysis – and further application of state of the art methods like this is needed.

185. The valuation of health endpoints in CBA in the WSS sector is partial. The coverage of health impacts is often limited to fatal and non-fatal diarrhoea. Even in these cases, cost-of-illness methods are used for morbidity, which do not consider pain and suffering.

186. Further studies on health valuation are needed, particularly in the developing country context – both for morbidity and mortality. In the meantime, before unit values can be established based on primary studies, systematic reviews or meta-analysis of a wide range of studies in the developing country context could provide the basis for mortality valuation. Forthcoming OECD work in this area may provide a useful basis for this. For morbidity, further primary studies are needed in developing countries before this is possible – the evidence base is simply too weak. For this case, it is recommended that a range of values based on transfer from studies in other countries is used until more primary studies become available.

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