WHO/SDE/WSH/00.10
English only
Distri: Limited

Considerations in evaluating the cost-effectiveness of environmental health interventions

Protection of the Human Environment
Geneva, 2000

CONTENTS

LIST OF TABLES

LIST OF BOXES

FOREWORD

This document has been developed with the aim to provide an overview of currently used methods for economic evaluation and to discuss implications of using these methods for evaluating environmental health interventions. It aims at formulating recommendations for future evaluations in environmental health. The document is intended to contribute to the methodological discussions, and in particular the development of guidelines for evaluation of cost-effectiveness in the framework of WHO's initiatives, and other ongoing work in this area. This work constitutes a background document with preliminary considerations of methods for economic evaluations in environmental health.

This document in particular focuses on what is peculiar to environmental health interventions, and therefore how the conduct of economic evaluations may need to be different to other health interventions. The main peculiarity, or difference, is that environmental health is a cross-cutting area. Environmental health interventions may need to be addressed, funded and implemented by various sectors (the health sector, the environment sector, the industrial sector, the transport sector, water or infrastructure services). In return, the benefits from environmental health interventions also accrue to various sectors and sometimes also to fulfil basic needs and increase the comfort or quality of life of the receiver (such as improved water supply). This issue raises the question: "*Who will pay for (which part of) the intervention?*". In relation to economic evaluation, therefore, it should be decided how the benefits should be accounted for in the cost-effectiveness (or cost-benefit) ratio.

Furthermore, environmental health also deals with some exposures with very long-term effects, such as climate change, certain occupational exposures, changes in ecosystems with (short and) long-term effects on health, for example through the change in vector populations. Discounting health, even at very small rates, would make almost any impacts with very long latencies seem negligible, thus raising the question *"how can discounting for long-term health effects be compatible with the concepts of prevention and sustainability?"*

This document represents a work-in-progress, and comments are gratefully received:

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EXECUTIVE SUMMARY

The motivation for this review and discussion paper arose from the current development and pending publication of the WHO guidelines on cost-effectiveness analysis (CEA), and the interests of certain groups to feed into these guidelines. The influence and impact of such CEA guidelines is potentially considerable, due to the current gap in comprehensive cost-effectiveness guidelines that are applicable to a wide range of health interventions *in developing countries*; also, through the influence of the WHO in research and policy making in these countries, the WHO guidelines are guaranteed a wide usage. Within this context, the Department of Protection of the Human Environment (PHE), contained within the Cluster of Sustainable Development and Healthy Environments, commissioned this study, to examine the implications of the cost-effectiveness guidelines for health interventions related to changes in the environment. The Terms of Reference for this study mentioned, among other things: how non-health costs and benefits should be taken into account in cost-effectiveness analysis to reflect the efficiency of environmental health interventions; how regulatory mechanisms can potentially be evaluated using CEA guidelines; the optimal valuation methods for quantifying costs and benefits in monetary units; the appropriate interest rate for discounting future costs and benefits of environmental health interventions; and how to deal with the uncertainty surrounding the cost-effectiveness of environmental health interventions.

Several important findings have arisen from this review. The first finding is that there is a serious lack of cost-effectiveness studies for all types of environmental health interventions, and therefore decision makers have limited information on the relative cost-effectiveness of health interventions from which to make evidence-based decisions. Also, there is lack of clarity in the current literature about which methods should be used for evaluating environmental health interventions. The second finding is that the Ministry of Health is unlikely to consider the costs and benefits arising to other agents or ministries, despite the importance of these cost and benefits arising from many environmental health interventions. However, the Ministry of Health could be persuaded to include costs and benefits which have implications for the financing or implementation of these interventions. The implication of this is, however, that when adopting the Ministry of Health perspective in evaluating cost-effectiveness, the true efficiency of many environmental health interventions is not measured, resulting in a cross-sectoral misallocation of resources. One possible, although data-intensive, solution proposed is to first define a range of perspectives (each one containing different types of cost and benefit), second to collect data for all relevant perspectives and present them separately, and finally, and third leave it for politicians to decide which viewpoint to adopt for decision making.

The third finding is that the valuation methods for valuing non-health environmental benefits in monetary units are underdeveloped, especially for application in developing country settings. However, there is a base of research to start with and further research can draw on findings in the economics literature. The fourth finding is that there is widespread disagreement about what discount rate should be applied to environmental projects. Although a number of solutions are proposed, it is recommended to use the positive discount rates given in CEA guidelines, as well as a 0% discount rate, and again leave it to the politicians to decide. The fifth finding is that the impact of environmental health interventions in terms of cost-effectiveness is highly uncertain, due to methodological difficulties, lack of reliable data and non-generalisability of data between settings. Therefore, once the CEA framework is defined for evaluating environmental health interventions, further research should be commissioned not only in collecting primary data but also in adapting the results to increase relevance for decision makers in a range of settings.

In conclusion, this document has served to pinpoint critical issues in the economic evaluation of environmental health intervention, it has proposed a range of solutions, and discussed their appropriateness from a range of viewpoints. In particular, possible problems that may occur in applying currently used CEA guidelines to the economic evaluation of environmental health interventions are raised, and solutions proposed. In formulating the WHO CEA guidelines, these issues should be taken seriously, due to the wide range of health risks and hazards from environmental sources affecting large parts of the world population. Where satisfactory solutions for those working in the field of environmental health cannot be agreed, it is recommended that special provisions are drawn to allow a ‘fair’ evaluation of environmental health interventions. However, this document is only seen as one of many documents and viewpoints that will feed into the final WHO CEA guidelines.

ACKNOWLEDGEMENTS

I would like to express my gratitude to several people who have helped in the formulation and writing of this document. First, I would like to thank members of the Department of Protection of the Human Environment at WHO who have been proactive in making sure that the viewpoint they represent is not ignored or passed over in the development of the WHO cost-effectiveness guidelines. Without them and their funding support, this document would never have come about. In particular, my thanks go to Jamie Bartram, Robert Boss, and Annette Pruess, all of whom gave valuable comments on the first draft of this document.

Second, there are several individuals representing European agencies connected to the environment and occupational health who provided valuable documentation, which allowed this review to be comprehensive in the short time available for the work.

Third, I would like to thank Anne Mills (Health Policy Unit, London School of Hygiene and Tropical Medicine, UK) for her initial help in getting this work started, and her comments on the first draft of this document.

Finally, I would like to thank members of the Centre for Social and Economic Research on the Global Environment, UK (in particular, Stavros Georgiou and Brett Day) for their input to locating important economic literature during the early stages of the work.

ABBREVIATIONS AND ACRONYMS

(see TABLE KEYS for abbreviations used in tables)

AE	Avertive expenditure
BMJ	British Medical Journal
CBA	Cost-benefit analysis
CE	Cost-effective
CEA	Cost-effectiveness analysis
CER	Cost-effectiveness ratio
COI	Cost of illness
CVM	Contingent valuation method
DALY	Disability-adjusted life year
DRF	Dose-response function
HC	Human capital
HPM	Hedonic pricing method
HRQL	Health-related quality of life
LDC	Developing country
MC	Medical costs
OSH	Occupational safety and health
PFM	Production function method
QALY	Quality-adjusted life year
RAD	Restricted activity day
RCT	Randomised controlled trial
TCM	Travel cost method
UK	United Kingdom
USA	United States of America
VOSL	Value of a statistical life
WLD	Work loss day
WTP	Willingness to pay

1. INTRODUCTION

The overall aim of health economic study from a public perspective is to make optimal use of resources expended in the health sector. 'Optimal' refers to meeting both efficiency (maximum health gains) and equity (appropriate distribution of health gains) objectives[1]. Therefore, resource allocation decisions should ideally be made on the basis of the comparative costs and consequences of all available interventions that affect health. However, very limited numbers of good quality cost-effectiveness or cost-benefit studies are currently available (Gerard et al 1992, Udvarhelyi et al 1992, Elixhauser et al 1998), especially in the developing world (Walker and Fox-Rushby 1998). In order to improve the quality and comparability of such studies, the World Health Organization (WHO) is currently drafting guidelines for conducting cost-effectiveness analyses on health interventions (Murray et al 2000)[2].

In principle, these new WHO guidelines should be comprehensive in scope, and therefore suitable for all types of interventions aimed at sustaining and improving health. There is a risk, however, as in the case of other widely used cost-effectiveness guidelines (Weinstein et al 1996, Drummond and Jefferson 1996), that their scope may be confined to include only those interventions typically delivered by core health services, with an emphasis on curative treatment. On the other hand, it should be recognised that many important health interventions are preventive in nature, some of which affect health through changes in environmental and social conditions.

Environmental health interventions differ from core health services in a number of ways. First, environmental health interventions are often regulatory in nature, acting on the fundamental cause of an injury or illness. Thus they are almost exclusively preventive, but their benefits may not be realised until the distant future. Second, environmental health interventions potentially convey considerable non-health benefits, such as saving time, increasing amenity, etc., which should be included when the viewpoint of the study is the societal one. Third, the primary responsibilities of funding and implementing environmental health interventions are often outside the domain of the health sector, and thus they require the collaboration or support of other sectors and/or ministries in their implementation. Fourth, the effectiveness of environmental health interventions is more difficult to evaluate than many core health services, as they are less amenable to controlled experiments due to the long time periods involved or they impact potentially large population groups but often in small amounts. Fifth, environmental health interventions also hold gains already achieved, and prevent 'back-sliding', which is often not taken into account in evaluation.

It is perhaps for these and other reasons that selective primary health care interventions, such as those suggested in the influential article by Walsh and Warren (1980), contain limited environmental health interventions, and those environmental health interventions included are concluded to be much less cost-effective than most curative measures. Several recent attempts have been made to formulate essential

[1] Note that a trade-off between these two sometimes opposing objectives may be necessary.
[2] The WHO programme "Choosing interventions: cost, effectiveness quality and ethics", part of the Global Programme on Evidence for Health Policy, is attempting to address some of the challenges of providing decision makers with timely information on the technical and ethical characteristics of different interventions to inform health policy debates (Murray et al 2000).

national packages of services in developing countries, few of which contain environmental health interventions. For example, the set of interventions considered by Bobadilla et al (1994) for developing countries contained no health interventions that could be described as 'environmental' in nature. On the other hand, Jha et al (1998) included the construction of pit latrines and safe water provision as part of a package of forty health interventions in Guinea, but this intervention turns out to be considerably less cost-effective than the treatment of diarrhoea. Others have argued, however, that environmental health interventions to prevent diarrhoea can compete with other means of controlling diarrhoea, such as oral rehydration therapy, once the special factors that make environmental health interventions different from curative health interventions are taken into consideration (Varley 1998). Economic evaluation issues with respect to water, hygiene and sanitation issues are discussed in a chapter of a forthcoming book (Hutton 2001).

Therefore, it could be argued that the stance of many ministries of health in formulating essential health packages, which are heavily influenced by the international literature and policy advice of 'outsiders', does not accord with all of the strategic policy directions adopted by the WHO.'s global cabinet, two of which are relevant to environmental health (numbers 2 and 4 in Box 1.1 below).

Box 1.1: Strategic policy directions adopted by the WHO's global cabinet

1. Reducing the burden of excess mortality and disability, especially that suffered by poor and marginalised populations.
2. Reducing the risk factors associated with major causes of disease and the key threats to human health that arise from environmental, economic, social and behavioural causes.
3. Developing health systems which are managed to ensure equitable health outcomes and cost-effectiveness; responsiveness to people's legitimate needs; are financially and procedurally fair; and encourage public involvement.
4. Promoting an effective health dimension to social, economic and development policy.

For mainly the reasons discussed above, the special aspects of environmental health interventions need to be considered when conducting economic evaluations, although it should be noted that not all environmental health interventions fall within these descriptions above, nor are these attributes unique to environmental health interventions. In the light of the impact of the above differences with core health services, conclusions need to be made concerning the appropriateness of the WHO draft guidelines on cost-effectiveness, and the extent to which they need adapting to allow fair comparison of environmental health interventions with other types of health intervention. The theoretical and practical feasibility of this adaptation to accommodate environmental health interventions must also be questioned. If difficulties are experienced, the possibility should be recognised that environmental health interventions need to be evaluated in a different framework to other health interventions.

The terms of reference for this review suggest several issues for consideration. These issues include:

1. Environmental health interventions involve benefits besides health benefits, such as greater amenity and saved time in the case of improved water availability, and avoided work loss and medical expenses in the case of water quality improvement.

How could these non-health benefits be taken into account in cost-effectiveness analysis to reflect overall benefits of environmental health interventions?

2. Most environmental health interventions are preventive. Thus costs that would have been incurred in the future due to injury or illness are 'saved'. *How can avoided costs due to prevented illness or avoided disease burden be taken into account in cost-effectiveness analysis?*

3. Funding of environmental health interventions from the health sector is potentially difficult in many countries, as changes to the environment are seen as the responsibility of other government departments, industry or landowners. However, people benefiting from the interventions may be willing to pay for them, not only citizens but also other government departments and private enterprise[3]. *First, how can this willingness to pay be taken into account? Second, should costs outside the health sector be included in the numerator of the cost-effectiveness ratio?*

4. In order to increase comparability of the cost-effectiveness of health interventions, non-health benefits need to be converted into a single unit of measurement, such as monetary units. However, there exists considerable uncertainty about optimal methods for valuing these benefits, particularly as there is often inconsistency between valuation methods. *Therefore, how can non-health benefits be valued in monetary units?*

5. The impacts of environmental health interventions are observed after a relatively long period compared to curative interventions, disadvantaging them in terms of cost-effectiveness when benefits are discounted. Another important issue with respect to time is that environmental health interventions tend to be a more permanent solution to causes of ill heath, and therefore while they require a large initial investment the longer term intervention costs are less than curative services. Discounting future benefits, however, discourages sustainability. *Therefore, is discounting appropriate, and if so, what discount rate is appropriate for environmental health interventions?*

6. In estimating the cost-effectiveness of environmental health interventions, there is considerable uncertainty in some variables, as evidence may not exist and high quality data is expensive to collect. For example, Arrow et al (1996) wrote that cost-benefit analysis cannot be used for evaluating environmental, heath and safety regulations because there is simply too much uncertainty in the calculations. On the other hand, even economic evaluations based on uncertain data still have value, and can suggest whether health interventions are likely to be cost-ineffective, highly cost-effective, or somewhere in-between (Murray et al 2000). In all forms of economic evaluation, it is important to be clear what uncertainty exists such as listing assumptions and providing ranges on values. *Therefore, how can sensitivity analysis*

[3] In fact, there are many examples of collaboration between the health sector and other government departments or industry to achieve an environmental improvement, such as in laying water and sewerage pipes, reducing air pollution in cities, or reducing the spread of pathogens via food. Traditionally, the health sector may not be responsible for many of the costs involved. However, in a decentralised government system this means that the health sector has less influence in whether such projects go ahead, although they are of major interest to the health sector.

be used to quantify this uncertainty? What are the weaknesses of sensitivity analysis in quantifying uncertainty in environmental health interventions?

7. The environment sector and the discipline of economics have also had to deal with these six issues listed above. *Are there lessons to be learned from solutions proposed in these fields?*

The terms of reference also mentions briefly the types of environmental health interventions that should be included in the review. In such a review of the economics of environmental health interventions, naturally a definition of an ‘environmental’ health intervention is required, although definitions are not easy to find in the literature. A comprehensive definition may be “projects that involve making changes to the natural or human environment or to human behaviour that have beneficial impacts (or prevent adverse impacts) on the health of humans”. As guidance, the terms of reference states that environmental health interventions include air pollution, water supply and sanitation, radiation, climate change, food safety, exposure to chemicals, noise, water resource management and vector control, etc. Therefore, the review includes these interventions, but also with the addition of occupational safety and health (OSH) interventions, as well as waste management interventions, which are both clearly improvements to the human environment. Another improvement, that of housing, is partially covered within air pollution (indoor air quality), water supply and sanitation, and waste management, and therefore will not be covered separately.

In order to place these environmental health interventions in context, Table 1 (taken from WHO 1997) shows potential relationships between exposure situations and/or positive hygiene behaviour and some of the most important health conditions in terms of global burden of disease (Murray and Lopez 1996). It is clear that the six categories of environmental hazard have significant adverse effects on human health, thus suggesting that reductions in these hazards have considerable potential for improving human health. While WHO (1997) attempts to quantify the extent to which environmental health interventions could alleviate these disease burdens, they admit the difficulties in such calculations. For example, polluted water and contaminated food are jointly responsible for roughly 90% of diarrhoeal diseases, and therefore basic sanitation, water supply and food safety interventions are likely to prevent a large proportion of these.

Table 1.1. Potential relationships between exposure situations and health conditions

Health conditions of concern	**Polluted air**	**Excreta and household wastes**	**Polluted water or deficiencies in water management**	**Polluted food**	**Unhealthy housing**	**Global environmental change**
Acute respiratory infections	•				•	
Diarrhoeal diseases		•	•	•		•
Other infections		•	•	•	•	
Vector-borne diseases		•	•		•	•
Injuries and poisonings	•		•	•	•	•
Mental health conditions					•	
Cardiovascular diseases	•					•
Cancer	•		•	•		•
Chronic respiratory diseases	•					•

(from WHO 1997, page 132)

To place in context the impacts of development on the extent of environmental health hazards, Table 1.2 distinguishes between "traditional hazards" associated with lack of development, and "modern hazards" associated with unsustainable development (WHO 1992). WHO (1997) points out a useful distinction between traditional hazards and modern hazards in terms of the speed with which they are expressed as disease: whereas traditional hazards generally have an immediate impact, modern hazards build up over time, making it difficult to prove the impact on health status.

Table 1.2. Main location and types of hazards in 'traditional' and 'modern' societies

Medium or location	**"Traditional hazards"** (*developing countries mainly)*	**"Modern hazards"** (*developed countries mainly)*
Water, food and sanitation	• Lack of access to safe drinking-water • Food contamination with pathogens • Disease vectors breeding • Inadequate basic sanitation • Drinking water pathogen outbreaks	• Water pollution from populated areas, industry and intensive agriculture • Food additives and modern preservation techniques
Air	• Indoor air pollution from 'dirty' fuel • Urban air pollution from motor cars, coal power stations and industry	• Building materials and paints/solvents • Urban air pollution from motor cars, coal power stations and industry • (Re)-emerging infectious disease hazards
Workplace	• Biological, chemical, radiation, mechanical and physical hazards (agriculture and cottage industries)	• Chemical, radiation, mechanical and physical hazards (e.g. production lines, making 'modern' products)
Other outdoor environmental	• Inadequate solid waste disposal • Road traffic accidents • Natural disasters, including floods, droughts and earthquakes	• Solid and hazardous waste accumulation • Deforestation and land degradation • Climate change and stratospheric ozone depletion • Road traffic accidents

(adapted from WHO 1997)

Tables 1.1 and 1.2 are a starting point to formulating solutions to environmental health risks. The variety of ways in which the environment can be improved is presented below in Table 1.3, with examples of environmental health interventions alongside other health interventions.

The aims of this paper are therefore 2-fold:

1. To review studies evaluating economic aspects of environmental health interventions. These include empirical, methodological and review papers.
2. To discuss the implications of the WHO draft guidelines for the cost-effectiveness of environmental health interventions, and suggest necessary changes or increased emphases that are required to allow a ‘fair’ comparison of environmental health interventions with other health interventions. In particular, to assess the opportunities and obstacles to mainstream environmental health interventions into economic evaluation of health interventions, while also serving the links with other sectors.

In order to achieve these aims, the rest of this paper is structured as follows. Chapter 2 outlines current CEA guidelines[4], to be used as later reference when deciding how WHO draft CEA guidelines may need to be developed so that they can be applied fairly to evaluate environmental health interventions. Chapter 3 provides an overview of the environmental health economic literature. Chapters 4 to 8 are devoted to the five main issues mentioned above: benefit inclusion: cost inclusion; valuation; discounting; uncertainty. The final chapter, Chapter 9, draws conclusions about implications for cost-effectiveness methodology in evaluating environmental health interventions.

[4] At the time of writing, the WHO draft CEA guidelines were not complete, except generic costing spreadsheets and a concept paper (Murray et al 2000). Therefore, other widely accepted and used guidelines were used to elaborate the current CEA methods.

Table 1.3. Examples of environmental (and other) health interventions

Type	**Environmental health interventions**	**Other health interventions**
Water, hygiene and sanitation	Improved water quality[1] Improved water supply[2] Reduced use of polluted waters[3] Improved sewerage facilities[4] Hygiene education (e.g. hand washing)	Oral rehydration therapy Antibiotics Breastfeeding Safe weaning practices
Food safety and contents[5]	Improve food production Improve storage Improve packaging Reduce harmful substances Adequate cooking Hand and surface area washing Increase access to emergency treatment of poisonings.	Antibiotics Special diet following contamination
Vector-borne disease control	Environmental manipulation Environmental modification Personal protection Biological control Chemical control	Chemoprophylaxis Vaccination Bed nets Case detection Health education
Waste disposal	Appropriate storage before collection Regular and safe collection Appropriate storage in dump Safe incineration (smoke/water hazards) Well managed landfill	Treatment of ill effects of contact with waste using antibiotics or emergency treatment
Housing conditions	Safe building materials Ventilation Adequate size	Treatment of ill effects of poor housing, such as mental stress
Air pollution	Reduce use of fuel Reduce harmful substances in human environment Reduce human exposure to harmful air[6]	Emergency treatment Asthma inhalers
Climate change	Reduce emission of greenhouse gases Preventive measures to reduce harmful effects of climate change[7]	Emergency relief
Stratospheric ozone depletion	Reduce emissions of harmful gases Reduce human exposure to radiation	Treat cancers
Occupational safety	Reduce harmful exposures[8] and mental stress Improve social environment and job security	Treat illnesses and injuries
Noise pollution	Noise laws and law enforcement (neighbourhood noise) Exhaust silencers on cars	Treat illnesses and injuries

NOTES TO TABLE 1.3

[1] Water quality improvement methods vary according to cause of pollutant, concentration, location and cost. Main causes of water pollution are untreated sewage, nutrients causing eutrophication, synthetic organics (e.g. dioxins), acidification, and chemicals (e.g. arsenic, pesticides) (WHO 1997)
[2] Improved availability (piped water to house, standing well, improved haulage) as well as lower prices
[3] Education campaigns to stop people drinking polluted water or recreational use of polluted water
[4] Building pit latrines or installing sewerage pipes
[5] Main purpose of regulations is to reduce viruses, bacteria, parasites and toxins in food, but also increase food quality, such as increasing nutrients, vitamins, etc.
[6] Such as moving industry to less populated areas, improving ventilation in houses.
[7] Such as increasing efforts to control vectors, prevent food shortages, prevent flooding.
[8] Chemicals, radiation, biological agents, repetitive strain, mechanical dangers, etc

2. OUTLINE OF CURRENT COST-EFFECTIVENESS ANALYSIS GUIDELINES

Before discussing the issues in evaluating the cost-effectiveness of environmental health interventions, it is important to understand what CEA guidelines in current use recommend. Although the WHO draft guidelines on cost-effectiveness analysis were not complete at the time of writing, it is assumed they are not a significant departure from guidelines such as those published by the British Medical Journal (BMJ) Economic Evaluation Working Party (Drummond and Jefferson, 1996), and the recommendations of the United States Panel on Cost-Effectiveness in Health and Medicine (Weinstein et al, 1996)[5]. The BMJ guidelines are more comprehensive in that they cover all types of economic evaluation (i.e. cost-benefit as well as cost-effectiveness analysis), while the Weinstein et al guidelines use a purely cost-effectiveness framework. In addition to these, costing spreadsheets developed for the WHO guidelines were available, which give some indication of the levels and types of cost being included.

Economic evaluation guidelines have at least the three following aims:

1. To state clearly the principles of CEA and clear up misunderstandings.
2. To increase consistency and to allow comparison of different studies.
3. To clarify the methodological choices that can be made at various stages of the evaluation (such as whether to include certain costs and benefits).

For example, the Weinstein et al (1996) guidelines develop a set of recommendations for the practice of CEA that serve as a point of reference for investigators who seek comparability with other analyses in the literature (termed the ‘reference case’). The BMJ guidelines similarly aim to improve the quality of economic evaluations by agreeing acceptable methods and their systematic application before, during, and after peer review. However, the BMJ guidelines differ from the Weinstein et al (1996) guidelines in that they do not provide a reference case. The BMJ guidelines, on the other hand, provide a checklist of important quality aspects of economic evaluations, which is summarised below in Box 2.1.

Box 2.1 Summary of BMJ economic evaluation guidelines (from Drummond and Jefferson 1996)

Study design

(1) Study Question

-The economic importance of the research question should be outlined.

-The hypothesis being tested, or question being addressed, in the economic evaluation should be clearly stated.

-The viewpoint(s) – for example, health care system, society - for the analysis should be clearly stated and justified.

(2) Selection of alternatives

-The rationale for choice of the alternative programmes or interventions for comparison should be given.

-The alternative interventions should be described in sufficient detail to enable the reader to assess the relevance to his or her setting – that is, who did what, to whom, where, and how often.

[5] According to Murray et al (2000) these two guidelines represent current CEA practice. However, the WHO guidelines will take a different view to these, in that the average (ACER) instead of the incremental cost-effectiveness ratio (ICER) is measured. Murray et al (2000) state that the ICER is good for those making decisions at the margin, but it does not necessarily lead to the maximisation of health in society, as a reallocation of resources between health interventions may produce greater health gains. This they term the ‘sector-wide approach’.

(3) Form of evaluation
-The form(s) of evaluation used – for example, cost minimisation analysis, cost-effectiveness analysis (CEA) - should be stated.
-A clear justification should be given for the form(s) of evaluation chosen in relation to the question(s) being addressed.

Data collection
(4) Effectiveness data
-If the economic evaluation is based on a single effectiveness study – for example, a clinical trial – details of the design and results of that study should be given – for example, selection of study population, method of allocation of subjects, whether analysed by intention to treat or evaluable cohort, effect size with confidence intervals.
-If the economic evaluation is based on an overview of a number of effectiveness studies details should be given of the method of synthesis or meta-analysis of evidence – for example, search strategy, criteria for inclusion of studies in the overview.

(5) Benefit measurement and valuation
-The primary outcome measure(s) for the economic evaluation should be clearly stated – for example, cases detected, life years, quality-adjusted life years (QALYs), willingness to pay.
-If health benefits have been valued details should be given of the methods used – for example, time trade off, standard gamble, contingent valuation – and the subjects for whom valuations were obtained – for example, patients, members of the general public, health care professionals.
-If changes in productivity (indirect benefits) are included they should be reported separately and their relevance to the study question discussed.

(6) Cost data
-Quantities of resources should be reported separately from the prices (unit costs) of those resources.
-Methods for the estimation of both quantities and prices (unit costs) should be given.
-The currency and price date should be recorded and details of any adjustment for inflation, or currency conversion, given.

(7) Modelling
-Details should be given of any modelling used in the economic study – for example, decision tree model, epidemiology model, regression model.
-Justification should be given of the choice of the model and the key parameters.

Analysis and interpretation of results
(8) Adjustment for timing and costs of benefits
-The time horizon over which costs and benefits are considered should be given.
-The discount rate(s) should be given and the choice of rate(s) justified.
-If costs or benefits are not discounted an explanation should be given.

(9) Allowance for uncertainty
-When stochastic data are reported details should be given of the statistical tests performed and the confidence intervals around the main variables.
-When a sensitivity analysis is performed details should be given of the approach used – for example, multivariate, univariate, threshold analysis – and justification given for the choice of variables for sensitivity analysis and the ranges over which they are varied.

(10) Presentation of results
-An incremental analysis – for example, incremental cost per life year gained - should be reported, comparing the relevant alternatives.
-Major outcomes – for example, impact on quality of life - should be presented in a disaggregated as well as an aggregated form.
-Any comparison with other health care interventions – for example, in terms of relative cost-effectiveness – should be made only when close similarity in study methods and settings can be demonstrated.
-The answer to the original study question should be given; any conclusions should follow clearly from the data reported and should be accompanied by appropriate qualifications or reservations.

The following discussion identifies important issues that arise from these guidelines, with focus on issues most relevant to the economic evaluation of environmental health interventions. These issues include discussion of perspective (i.e. viewpoint) and which costs to include in the numerator and which effects to include in the denominator in the cost-effectiveness ratio, definition of economic value, source of effectiveness evidence, the use of models, valuation of benefits, discounting, and uncertainty.

Perspective and which items to include in the numerator and denominator
Guidelines recommend the adoption of a societal perspective, defined as the costs and benefits (except transfers) that fall on all agents affected by the intervention. Guidelines suggest that costs and benefits to different agents should be provided separately as well as in aggregate, so that the analysis can be carried out from a number of viewpoints (e.g. health care system, third party payer, patient). In order to maximise the usefulness of the research, investigators should identify key potential decision makers at the outset of the study and be able to show that the research question posed will meet the needs of all key groups.

Weinstein et al (1996) argue that the formulation of the cost-effectiveness ratio in which net expenditure of health sector resources (a monetary measure) goes in to the numerator and the net improvement in health (a non-monetary measure) goes into the denominator, is incomplete for two main reasons:

- It leaves open to question whether certain costs and consequences should be thought of as health care costs or savings (numerator) or health decrements or improvements (denominator).
- It ignores non-health costs and effects, such as those associated with productivity, the environment or education.

Therefore, the variables that should be included in the denominator and numerator are discussed in turn.

The denominator: In CEA, this should only capture the health effect of the intervention, secondary as well as primary health effects. In order to avoid double counting, monetary values for lost life-years should not be included in the numerator of the cost-effectiveness ratio (CER), as the health effect is already in the denominator. However, the health-related quality of life (HRQL) measure in the denominator should implicitly incorporate the effect of morbidity on productive time[6]. Therefore, according to the recommendations of Weinstein et al (1996), valuation of

[6] This suggests that people who become more productive due to the change in health status will value their health-related quality of life higher.

HRQL in monetary terms would not constitute a reference case analysis[7]. The value of time spent sick (morbidity time) is part of the health outcome, and should be included in the denominator, although there are times when it could be included as an input to the health care process such as time spent recuperating from surgery.

<u>The numerator:</u> This should capture changes in resource use associated with an intervention, including costs of health services, costs of patient time expended for the intervention, travel expenses, costs associated with care giving (paid or unpaid), child care, and economic costs borne by employers, other employees and the rest of society (Weinstein et al, 1996). These latter categories include so-called friction costs associated with absenteeism and employee turnover, and costs to the educational system, the criminal justice system, or the environment. Costs associated with health care that does not affect outcomes, such as research costs, should not be included. Costs averted due to an intervention should be subtracted from the cost estimates, to give net cost. These include medical costs saved as well as costs due to avertive behaviour saved. [However, cost savings to other sectors in the economy are rarely included in the numerator of the cost-effectiveness ratio.]

Weinstein et al (1996) take special care to distinguish between which future health care costs (of people affected by the intervention) should be included. One distinction is made between health care costs for diseases related to the intervention and those not related to the intervention, where the former is included and latter excluded. Another distinction is made between those health care costs incurred in years of life that would have been lived anyway and years of life extended by the intervention, both of which are included in the reference case. A more controversial cost is that of diseases unrelated to the intervention that occur in added years of life. The reference case can either include or exclude these, and sensitivity analysis done if it is thought the alternative scenario may change the results significantly. A final category of consequential costs are non-health care costs in added life-years, such as general consumption like food, clothing, shelter, etc. While the economic literature often argues to include the net economic burden of survivors (consumption minus productivity) as a cost, the reference case does not include these costs, thus avoiding the controversial issue of how to ration resources between productive and non-productive people.

Economic value

Current guidelines argue that prices should be used to approximate economic value except where distortions are likely to be significant and important to the analysis (Weinstein et al, 1996). This includes adjustment by cost-to-charge ratios to take out the profit element in private health care, and adjustment of costs by the inflation rate when costs are used from a different time period than that of the study. However, this raises the issue of how to identify distortions, and how to make appropriate adjustments, which is not adequately described by the guidelines, but has been addressed in more detail in the cost-benefit literature under the term 'shadow prices' (Little and Mirrlees 1982). For example, the value of time of individuals not already included in staff costs, whether health care providers or patients, should be valued by the wage rate as an acceptable measure of opportunity cost of time (age- and gender-

[7] The 'reference' case is a defined framework outlined by Weinstein et al (1996) which increases the comparability of cost-effectiveness ratios across different health interventions and studies.

specific). However, shadow prices can be highly context-specific, as they depend on current market conditions.

The issue of what constitutes an opportunity cost has also been addressed briefly by guidelines. For example, when comparing two or more interventions, costs and benefits should be measured at the margin and therefore reflect extra costs consumed or saved due to a given health intervention, rather than total resources. However, the Weinstein et al reference case involves using a long-term perspective which requires that costs fixed in the short-term should be included. The BMJ guidelines distinguish between studies relating to decisions of the hospital manager in the short-term, which should use marginal costs, and those relating to national policy, which should use average costs.

Source of evidence of effectiveness
Weinstein et al (1996) state that the quality and validity of a CEA depend crucially on the quality of the underlying data that describe the effectiveness of interventions and the course of illness without intervention. For the purpose of a reference case analysis, acceptable data for estimation of effectiveness may come from a variety of sources: randomised controlled trials (RCT), observational studies, uncontrolled experiments, or descriptive series. Outcome probabilities should be selected from the best designed (and least biased) sources that are relevant to the question and population under study. While economic evaluations alongside randomised, double-blind controlled trials are the gold standard for CEA as they contain the least bias, modelling and ad hoc syntheses of data from several sources including expert opinion is accepted as a valid and necessary scientific procedure for estimating effectiveness. Also, while meta-analyses of effectiveness data, such as a Cochrane review, provide even stronger evidence of the effectiveness of interventions than most single RCTs could, the methods for identifying cost-effectiveness using Cochrane reviews has not been fully developed yet (Vale et al 2000). Models are advised not to be habitually used as they are not substitutes for direct primary or secondary empirical evaluation of effectiveness. External validity of RCTs should also be assessed in using results in other settings, by first identifying whether the trial conditions mirror actual conditions, and second identifying differences in settings such as characteristics of health systems and populations, and make adjustments where relevant (although few studies address these issues).

Measure of health benefit
In CEA benefits are usually measured in natural units, and for a CER only a single measure can be used – either the most important health impact, or a single measure such as utility that incorporates several types of health impact. The reference case of Weinstein et al (1996) advocate use of quality-adjusted life-years (QALY), which incorporate changes in survival and changes in HRQL by weighting years of life to reflect the value of health-related quality-of-life (HRQL) during each year. The weights used in QALYs should be based on a generic health-state classification system that reflects health-related domains that are important for the particular analysis, and be based on community preferences rather than patients or investigators. When calculating QALYs gained from a life-extending intervention, estimates of age- and sex-specific HRQL should be applied to the years of extended life – even if the intervention itself has no effect on HRQL. However, the impact of socio-demographic characteristics on HRQL and therefore the relative effectiveness of an intervention on

different groups of people must be made clear, due to ethical problems of giving different people different HRQL for similar changes in health status.

In cost-benefit analysis the benefits of health care are traditionally valued in money terms by either using the human capital approach or the willingness to pay (WTP) approach. However, double counting may occur if these benefits are measured alongside another method of valuing improved health. Also, WTP is heavily influenced by ability to pay, with resulting bias towards providing services for the rich.

Modelling
Modelling may be required in many situations, including (Buxton et al 1997):
1. To extrapolate the progression of clinical outcomes beyond that observed in a trial.
2. To transform final outcomes from intermediate outcomes.
3. To examine the relation between inputs and outputs in production function models to estimate or apportion resource use.
4. To use data from a variety of sources to undertake a decision analysis.
5. To use evidence from trials, or systematic reviews of trials, to reflect what might happen in a different clinical setting or population.

The key requirements are that the modelling should be explicit and clear, as well as which variables/parameters have been modelled rather than directly observed in a particular sample, and main uncertainties noted.

Discounting of future impacts
The time horizon of the study should be long enough to capture all the differential effects of the alternative options. The reference case includes discounting of future costs and health outcomes occurring during different time periods to their present value, and states that they should be discounted at the same rate[8]. Weinstein et al (1996) suggest that a convention is needed for choosing the discount rate in order to achieve consistency across analyses. They argue that theoretical considerations suggest that the real discount rate should be based on time preference, which is the difference in value people assign to events occurring in the present versus the future. This is reflected in the rate of return on riskless, long-term securities, such as government bonds, which empirical evidence shows to be in the vicinity of 3% per annum. They claim that direct evidence on time preference for health outcomes is also consistent with a discount rate of 3%. While a 3% discount rate is the preferred rate for the reference case, it is also recommended to use 5% due to the large number of previous studies that have used this rate, as well as 0% and 7% in sensitivity analysis. The BMJ guidelines, on the other hand, suggest the analyst uses the government recommended rate, and conduct a sensitivity analysis using other rates. Also, the use of a zero discount rate for health benefits in the sensitivity analysis is suggested, so as not to penalise preventive programmes. The BMJ guidelines state that there is not enough empirical evidence on which to base a decision on the appropriate discount rate.

[8] Interest rates reflect people's preferences for having money and material goods sooner rather than later. According to Weinstein et al (1996) future health effects should be discounted at the same rate as future costs because people have opportunities to exchange money for health, and vice versa, throughout their lives. Failure to discount health effects will lead to inconsistent choices over time.

Handling uncertainty

CEAs are subject to uncertainty with regard to estimates of effectiveness, the course of illness, HRQL consequences and preferences, and health care utilisation and costs. Briggs et al (1994) recognised three categories of uncertainty:

1. Uncertainty relating to observed data inputs. Typically, confidence intervals might be presented, the size of which depend not only on sample size, but also on within sample variability.
2. Uncertainty relating to extrapolation. This includes data generalised from other settings, as well as data modelled using epidemiological models or regression.
3. Uncertainty relating to analytic methods. Clearly there are several choices for analytic method, while often there is no clear way of knowing which is the appropriate one to use, such as the discount rate. This is aided by the reference case given by Weinstein et al (1996).

Users of information need information on the degree to which CEA conclusions might change with changes in assumptions or values. Sensitivity analysis is recommended by all CEA guidelines as an appropriate tool with which to respond to this need. One-way sensitivity analysis (assessing the impact of alternative values or assumptions one by one) is generally useful to know the extent to which uncertainty in single variables affects the results. Multi-way sensitivity analysis is recommended to assess the extent to which the results vary when more than one source of uncertainty exists, and possible correlation (positive or negative) between sources. Probabilistic sensitivity analysis is a further refinement, which attaches probabilities to ranges, and thus allows a probability distribution of the cost-effectiveness ratio to be estimated.

Conclusion

Guidelines such as those discussed above are immensely powerful tools to change practices, especially when journal reviewers and editors use them to judge whether to publish economic evaluations or not, or similarly when funding agencies only fund economic studies that will meet minimum quality criteria. Guidelines also encourage standardisation, such as the ‘reference case’ attempts to do, which is viewed as a positive effect when cost-effectiveness ratios must be put together in league tables as an aid to decision making. Standardisation implicitly assumes that health interventions can be compared in a standardised way, which may disadvantage some health interventions that do not fall within the classification categories of core or curative health services, as discussed in Chapter 1. The guidelines discussed above are biased towards health sector impacts and therefore do not implicitly take a multi-sectoral perspective, nor therefore support inter-ministerial activities. Therefore, as stated in the aims, later chapters evaluate these guidelines from the perspective of environmental health interventions.

3. OVERVIEW OF ENVIRONMENTAL HEALTH ECONOMICS LITERATURE

3.1 Literature search and review strategy

A brief search was made of the environmental health economics literature, using key words in medical and development databases[9], following up references in collected articles, accessing the web sites of relevant agencies and university departments[10], and contacting of key agencies[11]. There was a diversity of economic studies found, ranging from studies that focussed on single economic variables, such as costs of implementing regulations and willingness to pay for environmental health services, to full economic evaluations. Also, a handful of publications reviewed economic studies or economic issues in environmental health.

The purpose of this section is 2-fold:

1. To list publications found in the literature search by environmental health topic, and present important aspects of each study, including study authors, aims, country of study, costs and benefits included, and data sources.
2. To (briefly) appraise these studies in terms of important economic evaluation quality indicators, and identify key publications for each environmental health topic (either good economic evaluations and/or studies that discuss important methodological issues).

However, it should be noted that due to constraints on time for conducting a systematic literature search, and because many publications identified in data bases could not be accessed in the review period, this was by no means a comprehensive review. On the other hand, some review articles were found on a few environmental health topics, and therefore the conclusions of these reviews are used. For example, Giorgiou et al (1998) conducted a review of economic valuation of environmental changes, including health impacts, and the annotated bibliography is used to identify relevant publications. First, general publications on environmental economics are reviewed briefly, followed by the review of economic studies of environmental health interventions.

3.2 Publications on environmental economics

A large number of books and articles were found on the subject of environmental economics, many of which contained references to health impacts, and valuation of health impacts (Freeman III 1979, Dixon et al 1986, Johansson 1987, Freeman III 1993, Hanley and Spash 1993, Beckerman 1995, Birley 1995, Fankhauser 1995, OECD 1995, Georgiou 1996, Field 1997). Some publications (e.g. Dixon et al 1986, Biswas and Agarwala 1992, Birley 1995, OECD 1995) focussed on environmental (and associated health) impact assessment of development projects, such as mining

[9] Key words were combined, such as 'cost', 'environment', 'health', 'air pollution', 'hygiene', etc. Databases included Medline, Popline, BIDS.
[10] Including USAID, World Bank, United Nations Environment Programme, Environmental Protection Agency, University of East Anglia/University College London (Centre for Social and Economic Research on the Global Environment), Leicester University Centre for Environmental Studies.
[11] US National Institute for Occupational Safety and Health; UK Tayside Occupational Health and Safety Service; TNO Work and Employment; European Agency for Safety and Health at Work; Eurostat; UK Health and Safety Executive; International Council on Metals and the Environment.

and power generation. These publications listed above contain discussions of many relevant topics to this review, such as valuation, uncertainty, discounting, etc, and are referenced later in the discussion.

3.3 Economic studies of environmental health interventions

The brief search of the literature found very few complete economic evaluations focussing on the cost-benefit or cost-effectiveness of environmental health interventions. Many studies that met the basic criteria for an economic evaluation[12] either did not use primary data sources for measurement of costs and benefits (e.g. Varley (1998) for WHS interventions), or they did not include all the costs and benefits that would be considered necessary for a decision maker to make a proper choice (e.g. Krupnick and Portney (1993) for urban air pollution, Hanley and Spash (1993) for nitrate pollution reduction). Also, there were several CEA or CBA publications found on the web page of the US Environmental Protection Agency (EPA) on groundwater protection, effluent emissions reduction etc. that could not be accessed in the time available.

In addition to these economic evaluations, there were a large number of studies that measured economic variables necessary for a full economic evaluation, and therefore these studies contributed to the discussion of methodological issues in conducting cost-effectiveness analyses on environmental health interventions. Most notable among these were willingness-to-pay studies by Whittington and others on water supply, and other studies measuring the cost of illness of food contamination, occupational injury and disease, and air pollution. Several publications reviewed published literature or economic issues, such as Mossink et al (1998) and Leight et al (1996) for occupational safety and health, UK Department of Health (1999) for air pollution, WASH (1991) for water and sanitation, Mills (1991) and WHO (1986) for vector control, and Postle (1997) for chemical risk management.

In summary, the environmental health interventions that were most represented in the economics literature in decreasing order were occupational safety and health; air pollution; environmental vector control; water, hygiene and sanitation; chemical management; climate change; food safety; and waste disposal. Health economic studies of some environmental health interventions were not found at all using the literature search strategy, including prevention of radiation, improved housing, and preventing ozone depletion. Below, the findings of economic studies for each environmental health intervention are discussed.

3.3.1 Water, hygiene and sanitation

There were a variety of studies found that evaluated the economics of water, hygiene and sanitation (WHS) interventions. Important aspects of these studies are summarised in Table 3.1. Costs are distinguished by who they are borne by, and benefits according to the method of valuation (see notes under table). No studies were found that estimated cost-effectiveness of WHS interventions using primary data from a single setting. One study, however, modelled the cost-effectiveness of WHS interventions using secondary data from a variety of sources (Varley et al 1998). This study estimated both average and incremental cost-effectiveness ratios. ACER was estimated for 3 options – hardware only, software only, hardware and software. Also,

[12] That is, comparing two or more alternatives and comparing costs and consequences.

Phillips (1993) discussed the potential cost-effectiveness of hand washing to prevent diarrhoea, using published studies of effectiveness data to build a plausible picture of procedures and resource use, and hence of costs. She comments that no cost-effectiveness studies were found on WHS interventions. Finally, Briscoe (1984) discusses methodological issues in evaluating the cost-effectiveness of WHS interventions, presenting data to support the hypothesis that WHS interventions can compete with oral rehydration in terms of cost-effectiveness in reducing the incidence of diarrhoeal diseases.

Table 3.1 shows a number of other studies that evaluated economic aspects of WHS interventions, although they were not comprehensive in evaluating and comparing the full costs and benefits of WHS interventions in a cost-effectiveness ratio. These studies were of three main types – those that evaluated the costs of WHS interventions, those that evaluated the cost-of-illness associated with diseases due to lack of WHS facilities, and those that measured the willingness to pay of improved water supply. However, very few studies included the costs of water and sanitation services, and those that did used crude data that was often transferred from other countries. The focus of other economic studies was on the benefits of improving WHS[13]. Several studies measured the cost of illness, most notably the USAID-sponsored water and sanitation for health (WASH) project which published methodological work on cost of illness (WASH 1991) as well as applying the methodology in the field (Paul and Mauskopf 1993). Also, Harrington et al (1989) measured both morbidity losses and costs of preventive actions to avoid water-borne giardiasis.

Generally the studies assessing the water market focussed on the demand side, such as willingness to pay and identifying options for cost recovery. Willingness to pay studies were of two main types: improvements in the availability and quality of drinking water, and improvements in the quality of ground water (rivers, lakes or coastal waters) for amenity uses. The literature of the former type of study was dominated by the work of Dale Whittington (World Bank). Most WTP studies used the contingent valuation technique to identify the potential demand curve for improved water supply and quality, and many of these also identified current water markets and compared them with WTP (Whittington et al 1990a, 1990b). Darling et al (1992) determined WTP for improved sanitation services in the Caribbean, to prevent decay to coral reef and allow bathing to continue. North and Griffin (1993) identified WTP using differences in house prices based on housing characteristics, which included water supply. However, none of these studies compared WTP with actual cost.

[13] Studies measuring the clinical effectiveness of WHS interventions are not discussed here, and reviews of these studies are found in Esrey et al (1985, 1991), Blum and Feachem (1983) and Feachem (1986). Uncertainties associated with the effectiveness of WHS interventions are discussed later.

Table 3.1 Economic studies on water and sanitation services

Reference	Study aim and country	Costs included	Benefits included
Cost-effectiveness or cost-of-illness studies			
Briscoe (1984)	Review of cost-effectiveness of water supply	R: HS	R: MOR
Harrington et al (1989)	Costs of a waterborne disease outbreak (USA)	P: HS, PT	P: COI
Paul and Mauskopf (1991)	Methodology for cost-of-illness studies	None	R: COI
Philips (1993)	Review of diarrhoea control (LDCs)	S: HS	S: CDA
WASH (1993)	COI of cholera epidemic (Peru)	None	P: COI
Varley et al (1998)	CE of WS interventions (LDCs)	S: HW/SW	S: CDA, DALY
WTP studies on water supply and sanitation services			
Boadu (1988)	WTP for water piped to households (Ghana)	None	P: WTP
Whittington et al (1990a)	WTP for water from village standposts (Haiti)	None	P: WTP
Whittington et al (1990b)	WTP for water piped to households (Nigeria)	S: PIP	P: WTP
Whittington et al (1990c)	WTP for water - vendor/kiosk/wells (Kenya)	None	P: WTP
Whittington et al (1991)	WTP for improved piped water supply (Nigeria)	P: VE, HW	P: WTP
Darling et al (1992)	WTP for sewerage facilities (Caribbean)	None	P: WTP
Whittington et al (1992)	Time to think in WTP valuations (Nigeria)	None	P: WTP
Hanley (1991)	WTP for reducing nitrate level of water (UK)	None	P: WTP
North and Griffin (1993)	Water supply and house prices (Philippines)	None	P: WTP
Whittington et al (1993)	WTP for improved WS services (Ghana)	P: HW	P: WTP
WTP, cost and cost-effectiveness studies on water quality improvement			
Dixon et al (1986)	Industrial waste water disposal (Philippines)	S: IND	None
Hanley (1989)	Costs of reducing nitrate pollution (UK)	P: IND	None
Hanley and Spash (1993)	Review of CB of controlling nitrate pollution	R: PC	R: WTP, CAV
Kwak and Russell (1994)	WTP to stop contaminating river water (Korea)	None	P: WTP
WHO (1994)	Review of cost recovery approaches for WS	S: GOV	None
Giorgiou et al (1996)	WTP to improve bathing water quality (UK)	None	P: WTP
Day and Mourato (1998)	WTP to improve river water quality (China)	None	P: WTP
Machado et al (1999)	WTP to improve bathing water quality (Portugal)	None	P: WTP

TABLE KEY: *Abbreviations*: CE – cost-effectiveness; WS – water and sanitation; WTP – willingness to pay; LDCs – developing countries; CB – cost-benefit. *Data type*: P – primary data; R – review; S – secondary data. *Costs included*: HS – health service; PT – patient; PC – pollution control; GOV – government; VE – private vendors; IND – industry; HW – hardware; SW – software. *Benefits included*: MOR – morbidity and mortality; COI – cost-of-illness; CAV – costs averted; CDA - cases and deaths averted; DALY – disability-adjusted life years saved.

3.3.2 Food safety

Several economic studies were found that evaluated the economics of interventions related to food safety and food poisoning, summarised briefly below in Table 3.2. Only two studies were found that measured both the costs and benefits of intervention(s) to treat food poisoning, and both of these focussed on poison control centres (Donald et al, 1996; Miller and Lestima, 1997). However, no economic evaluations were found on food safety regulations (i.e. prevention). There were two main types of other economic study. One type measured the costs associated with foodborne illnesses, usually focussing on a single outbreak such as salmonella (Cohen et aal, 1978; Levy and McIntire, 1974; Buzby and Roberts, 1997). The other type estimated the WTP of individuals to pay for poison control centres. No studies measured the costs of any types of regulation in the food industry, although Buzby and Roberts (1997) provide a detailed classification of cost components.

Table 3.2. Economic studies on food safety and treatment of food poisonings

Authors/year	Study aim and country	Costs	Benefits
Levy and McIntire (1974)	Costs of salmonellosis (USA)	None	P: COI (PT)
Cohen et al (1978)	Costs of salmonellosis (USA)	None	P: COI (PT)
Garthright et al (1988)	Costs of intestinal infections (USA)	None	P: COI
Donald et al (1996)	Relative CE of poison control centre (USA)	P: HS	P: MOR
Buzby and Roberts (1997)	Costs of foodborne illness (USA)	R: All	S: MT
Miller and Lestina (1997)	Cost savings of poison control centres (USA)	P: HS	P: MOR
Phillips et al (1997)	WTP for poison control centres (USA)	None	P: CV

TABLE KEY: *Data type*: P – primary data collected; R – review; S – secondary data collected. *Costs*: HS – health service. *Benefits*: COI – cost-of-illness; PT – patient only; MOR – morbidity and/or mortality; CV – contingent valuation.

3.3.3 Vector control

Although there is considerable literature on the effectiveness of reducing vector-borne diseases using environmental control measures (Sharma, 1991), until 1986 only a few studies have been published that evaluated their cost-effectiveness (Sharma, 1986). Also, several more dated cost-effectiveness studies were reviewed by WHO (1986), including Little (1972), Fultz (1976), Provost (1977), Sarhan et al (1981), Shisler and Shultze (1981), Shisler and Harker (1981), Chan (1985), and Fultz (1986)[14]. Alternative environmental health activities relating to vector control were compared in reviews conducted by Wernsdorfer and McGregor (1988) and Mills (1991). However, no primary studies have been found that compared all alternative vector control options in a comprehensive framework for comparative estimates of cost-effectiveness or cost-benefit, or that compared several vector control activities with medical or health care activities.

Several publications review economic issues in evaluating cost-effectiveness of environmental control measures. For example, WHO (1986) summarised cost-effectiveness studies and technical issues in environmental management and Bos (1991) discussed cost-effectiveness considerations in vector control. Other economic studies evaluated beneficial impacts of vector control, such as impact on work time (Picard and Mills 1992) or impact of land use on malaria transmission (Sawyer 1993).

[14] However, these articles were in specialist journals, and could not be accessed in time.

Table 3.3. Economic studies on environmental management of vectors

Authors/year	Study aim and country	Costs	Benefits
Little (1972)	CE of various vector control options (Americas)	*	*
Debord (1975)	CE of chemical & non-chemical management (USA)	*	*
Fultz (1976)	CE of ditching & draining pastures (USA)	*	*
Provost (1977)	CE of dike maintenance and larviciding (USA)	*	*
Sarhan et al (1981)	CE of various vector control options (USA)	*	*
Shisler and Shultze (1981)	CE of EM and insecticide (USA)	*	*
Shisler and Harker (1981)	Permanent versus temporary control	*	*
PAHO (1983)	CE of various vector control options (Cuba)	*	*
Chan (1985)	CE of EM and insecticide (Singapore)	*	*
Fultz (1986)	Permanent versus temporary control (USA)	*	*
WHO (1986)	Review of CE of malaria control using EM	R: HS	R: MOR
Wernsdorfer & McGregor (1988)	Review of issues in economic evaluation of malaria interventions	R: HS	R: COI
Bos (1991)	CE considerations in EM of malaria	R: HS	R: MOR
Mills (1991)	Review of the economics of malaria control	R: HS	R: COI
Picard and Mills (1992)	Impact of malaria on work time (Nepal)	None	P: PROD
Sawyer (1993)	Economics of change in land use (Brazil)	None	P: COI
PEEM (1997)	Guidelines for vector control	R: HS, PT	R: MOR, COI
Konradsen et al (1999)	Costs of malaria control (Sri Lanka)	P: HS	P: IND

TABLE KEY: *Data type*: P – primary data collected; R – review. *Costs*: HS – health service; PT – patient. *Benefits*: COI – cost-of-illness; MOR – morbidity and/or mortality; PROD – productivity loss averted; EM – environmental management. * means these studies were found in review articles but were not accessed, hence it was not known which costs and benefits were included.

3.3.4 Waste management

Very few economic studies were found on waste management, whether the waste is of household or industrial origin. Only one economic evaluation was found, relating to disposal of household waste (Powell and Brisson 1994). This study included in costs the external costs of waste disposal, including global pollution, air pollution, transport impacts, and leachate. In benefits valuation, the pollution displacement as a result of turning waste into electricity was included. Therefore, health effects of reduced waste hazards were not included. Several publications discussed economic issues concerning waste disposal, including chemical risk management (Postle 1997), and the economic theory of solid waste management in developing countries (Pearce and Turner 1994). Cheong (1995) examined the pricing of municipal solid waste disposal in Korea. Brisson and Pearce (1995) examined the transferability of benefits results (using WTP) between countries. However, no microeconomic studies were found that evaluated cost effectiveness of waste management options in a comprehensive framework, with particular focus on health as a benefit.

Table 3.4. Economic studies on waste management

Authors/year	Study aim and country	Costs	Benefits
Powell and Brisson (1994)	CBA of waste disposal (UK)	P: GOV	P: EGD
Pearce and Turner (1994)	Economics of SWM (LDCs)	R: GOV	None
Cheong (1995)	Pricing for municipal SWM (Korea)	P: GOV	P: Demand curve
Brisson and Pearce (1995)	Benefits transfer for disamenity	None	R: CV, HP
Postle (1997)	CBA framework for chemical risk management	R: PC	R: COI, WTP

TABLE KEY: *Data type*: P – primary data collected; R – review; S – secondary data collected. *Costs*: PC – pollution control costs; GOV – government costs. *Benefits*: EGD – electricity generation displaced; COI – cost-of-illness; HP – hedonic pricing; CV – contingent valuation.

3.3.5 Air pollution

Although only one study compared the costs and benefits of air pollution control in the same study (Krupnick and Portney, 1993), considerable attention has been given to the economics of air pollution control, particularly with respect to the benefits of pollution reduction. A UK Department of Health (1999) publication provides a thorough methodological overview of issues arising in valuation of health benefits of air pollution control, including improvements in quality of life, willingness to pay, and cost of illness. There were many other economic studies addressing the impact of air pollution, and associated economic issues. Seethaler (1999) compared the cost-of-illness resulting from traffic pollution in three European Union countries, using a comprehensive human capital methodology. However, health-related quality of life from air pollution was not measured in any of the studies listed below[15], nor were costs of air pollution reduction included, or issues discussed.

Pearce (1996) finds ‘remarkably’[16] few studies measuring indoor air pollution, and many of those that do fail to make a distinction between ambient concentrations and exposure. Smith (1988) suggests that in both the developed and developing world, the dominant source of exposure to particulate matter may be indoors. Gas cookers and environmental tobacco smoke are the most serious sources of indoor air pollution in developed countries, while burning of wood and biomass creates the most serious exposure problems in developing countries.

[15] Although UK Department of Health (1999) describes in Annex 3A alternative acceptable measures of HRQL associated with air pollution.

[16] Probably ‘remarkable’ given that indoor air pollution is potentially responsible for the majority of ill health associated with air pollution in developing countries (WHO 1997).

Table 3.5. Economic studies on air pollution reduction and health

Authors/year	Study aim and country	Costs	Benefits
Waddell (1974)	Costs of stationary-source air pollution	None	P: HS, SO, NU
Lave and Seskin (1977)	Benefits from air pollution abatement	None	*
Freeman (1979)	Benefits from air pollution abatement	None	*
Fisher (1981)	Costs of environmental pollution	None	R: HS, CV
Ostro (1983)	Work loss and morbidity (USA)	None	P: COI
Hall et al (1991)	Economic value of cleaner air (USA)	None	S: COI
Lesmes (1992)	Costs of passive smoking	None	R: COI
Krupnick and Portney (1993)	CBA of controlling urban air pollution	S: IND	S: COI
Ostro (1994)	Work loss and morbidity (Jakarta)	None	P: COI (RAD)
Duborg (1995)	Mortality costs of lead emissions (UK)	None	S: VOSL
Pearce and Crowards (1995)	Costs of particulate air pollution (UK)	None	P: COI
Pearce (1996)	Costs of air pollution (LDCs)	None	R: COI
Gerking and Stanley (199-)	Costs of air pollution	None	P: CV
Alberini (1997)	Costs of air pollution (Taiwan)	None	P: CV
Bartanova (1997)	CBA for setting air quality standards	None	R: CV
Navrud (1997)	Costs of air pollution (Norway)	None	P: CV
Seethaler (1999)	Costs of air pollution from traffic (Austria, France, Switzerland)	None	P: COI
Department of Health (1999)	Costs of air pollution (general)	None	R: HRQL, CV, COI

TABLE KEY: *Data type*: P – primary data collected; R – review; S – secondary data collected. *Costs*: HS – health service; IND - industry. *Benefits*: COI – cost-of-illness; CV – contingent valuation; VOSL – value of a statistical life; HRQL – health-related quality of life; RAD – restricted activity days; SO – soiling costs; NU – non-use values. * means these studies were found in review articles but were not accessed, hence it was not known which benefits were included.

3.3.6 Climate change and stratospheric ozone depletion

There were only a handful of studies that evaluated the health impact and associated economic impact of climate change and stratospheric ozone depletion, although climate change has become the subject of many economics books. There were no complete economic evaluations of climate change, although several studies discussed economic issues. In this case there is a large literature from economics. The costs of measures to reverse climate change and stratospheric ozone depletion were not cited in economic studies in the health literature, except rough estimates made by Hanley (1993). Several studies attempt to estimate damage costs of climate change and stratospheric ozone depletion (e.g. Fankhauser 1992). In general, potential benefits were identified by most studies of reversing climate change (i.e. damage costs averted). In this case, health benefits are outweighed by many economic benefits of averting further climate change.

Health impacts of climate change are hypothesised to come from many sources, and have been classified under direct and indirect (or less direct) according to whether the health impact results directly from a change in climate, such as heat-related deaths or respiratory effects. Potential indirect health impacts of climate are many, and include changes in world food supply, spread of vector borne disease to previously temperate zones due to temperature changes, change in transmission patterns due to increased precipitation, and contamination of fresh water supplies due to sea level rise (WHO 1997, pages 122-125). Damage costs were estimated in a variety of ways, although none were comprehensive.

Table 3.6. Economic studies on climate change and stratospheric ozone depletion

Authors/year	Study aim	Costs	Benefits
Fankhauser (1992)	Damage costs of climate change	None	S: DC (HS, non-HS)
Hanley (1993)	CBA of the greenhouse effect	S:IND	S: DC
Fankhauser (1994)	Costs of greenhouse gas emissions	None	S: DC (AG, HS, SLR)
Tol (1995)	Damage costs of climate change	None	P: DC (HS, non-HS)
Goldsmith and Henderson (1999)	Costs of climate change	None	R: DC

TABLE KEY: *Data type*: P – primary data collected; R – review; S – secondary data collected. *Costs*: HS – health service; IND - industry. *Benefits*: DC – damage costs; AG – agriculture; SLR – sea level rise.

3.3.7 Occupational safety and health

Although there were few comprehensive economic evaluations of measures to increase occupational safety and health (OSH), this topic has received considerable attention in developed countries[17], although very limited attention in developing countries. Two literature reviews, one from the USA (Leigh et al 1996) and one from the European Union (Mossink et al 1998), summarise a considerable amount of literature on benefits valuation of work safety measures. For example, Mossink et al (1998) present tables of different types of economic study, distinguished by European country. It is not the purpose of this review to present details of these studies, but the reader is referred directly to their studies. Also, Mossink et al (1998) describe a methodology for estimating the cost-effectiveness of preventive measures in the workplace.

Most industry studies gave cost-of-illness estimates for different categories of injury and disease. For example, Leigh et al (1996) found nine studies looking at the costs of occupational illness and injury, and roughly 50 studies valuing statistical life using WTP (from vehicle crashes, smoke detectors, air pollution, cigarette smoking, and differences in automobile prices based on safety features). Leigh et al (1996) also provided their own estimates of COI and WTP. Mossink et al (1998) found that most economic models in intervention studies at the company level consider the three elements of (a) the current cost-of-illness (before intervention), (b) prevention costs (intervention), and (c) benefits due to prevention (post intervention). Many of these studies did not measure exact costs of every accident or illness, but instead used an average cost 'per incident' from a sample period and population. Also, these studies mainly measured monetary costs and benefits, and none were found that measured HRQL.

[17] According to Mossink et al (1998) interest in CBA of OSH issues is steadily growing, with several initiatives of interest (1) European conference on costs and benefits of OSH (1997); (2) Economic impact survey, conducted in EU member states by the European Agency for Safety and Health at Work; (3) Work insurance system with premiums based on risk rating of companies; and (4) a study by European Statistics on Accidents at Work on the costs of accidents at work.

Table 3.7. Economic studies on occupational safety and health

Authors/year	Study aim and country	Costs	Benefits
Aaltonen and Soderqvist (1988)	Classification of costs and benefits (Sweden)	R: All	R: All
Winkel et al (1994)	Economics of change in production line (Sweden)	None	P: Time
Davis and Teasdale (1994)	Costs of work related accidents (UK)	None	P: AE, HC, CV
Leigh et al (1996)	Review of costs and benefits of OSH (USA)	None	R, P: COI
Fahs et al (1997)	Costs of occupationally related disease (USA)	None	P: COI
Postle (1997)	Issues in CBA and chemical risk management	R: IND, GOV	R: COI, CV
Mossink et al (1998)	Review of costs and benefits of OSH (EU)	R: PREV	R: CAV, AE
EASHW (1998)	Review of current CBA practices in EU countries	None	None

TABLE KEY: *Data type*: P – primary; R – review. *Costs:* GOV – government costs; IND – industry costs; PREV – prevention costs. *Benefits:* COI – cost-of-illness; CAV – costs averted; CV – contingent valuation; HC – health costs averted; AE – averting expenditures averted, EU – European Union.

3.3.8 Other

There were few economic evaluations or economic studies on other environmental health interventions. One such study was Soguel (1994) who measured benefits from traffic noise reduction in Switzerland. This study used willingness to pay to value individuals preferences. Another was Dubourg (1993) who measured the WTP of drivers to reduce the risk of road traffic accidents in the UK.

Table 3.8. Other economic studies of environmental health interventions

Authors/year	Study aim	Costs	Benefits
Dubourg et al (1993)	WTP to reduce risk of road traffic accidents (UK)	None	P: CV
Soguel (1994)	WTP for traffic noise reduction (Switzerland)	None	P: CV

TABLE KEY: *Data type:* P – primary. *Benefits:* CV – contingent valuation.

4. BENEFIT INCLUSION

4.1 Introduction

This chapter reviews which benefits of environmental health interventions could or should be included in cost-effectiveness analyses under a variety of perspectives (for example, health system, consumer, societal). First the full range of possible benefits of environmental health interventions is listed. In particular, the advice of current CEA guidelines with respect to the inclusion of benefits is compared with those benefits identified in the environmental health literature. Later sub-sections review benefits specific to each major area of environmental health intervention in turn. In so doing, several issues are addressed:

1. What is the possible impact of the inclusion/exclusion of health and non-health benefits on the CEA results? Is their inclusion consistent with maximising welfare?
2. To what extent have previous studies measured benefits comprehensively?
3. To what extent should the health care decision maker be interested in non-health benefits?
4. How should avoided costs due to health benefits be taken into account?

5. Which agencies would be willing or able to pay for these benefits, to contribute to costs of environmental health interventions? [While this question is not directly relevant for economic evaluation, it could be important for implementation of interventions and for multi-sector analyses]

The section makes preliminary conclusions about which benefits should be included in the CEA of environmental health interventions, and which should be excluded.

4.2 General issues

It is widely recognised that the inclusion of purely health benefits in an economic evaluation understates the potential positive impact of environmental health interventions. As Berman (1982) points out, cost-effectiveness comparisons tend to undervalue interventions that provide important outcomes other than the one being considered, and are thus "particularly inappropriate where programs produce a broad mix of benefits". Postle (1997) identifies four categories of potential benefit arising from changes in the environment that influence health:

1. Benefits to *individuals*, including reduced mortality and morbidity (acute, chronic).
2. Benefits to *production or consumption*, such as to crops, fisheries, or industry.
3. Benefits to *economic assets*, such as corrosion of materials, and property values.
4. Benefits to *environmental assets*, such as recreational or passive use

This classification does not take into account, however, the reductions in health sector costs due to reduced illness. Similarly, OECD[18] (1995) recognises productivity, health, amenity and 'existence' values of environmental goods. The primary focus of the data collection and analysis of any study will, however, depend on the primary purpose or impact of the environmental change. For example, environmental improvements may be aimed specifically at improving crop yields, and thus measurement of health outcome is secondary to the analysis, and thus will receive less attention from the analyst. On the other hand, inclusion of health benefits can be used as further justification for environmental improvements aimed at other outcomes. Some environmental improvements may even contribute to health risk management at no or minimum extra cost. Conversely, an environmental alteration may be detrimental to health, such as an industrial project (see Dixon et al (1986) for some examples). In this case, inclusion of the costs associated with health impacts should be included, as is done routinely in health impact assessments (Birley 1995). Therefore, in theoretical terms, irrespective of the primary objective of a project, it is important to identify all relevant and important benefits or impacts.

Table 4.1 below, assembled from a variety of sources, lists the various benefits arising to different agencies from health interventions, with particular focus on environmental health interventions. For the purposes of the discussion, three main impacts of health improvements can be distinguished:

- *Impact on health status (H)*, and the associated quality of life (health-related quality of life- HRQL) and life expectancy (LE) changes. Measurement of these impacts are well developed in economic evaluation (mainly CEA).
- *Impact on economic productivity (PR)*, due to reduced mortality and morbidity (improved HRQL and LE). Measurement of these impacts are also well developed in economic evaluation (mainly CBA).

[18] In the publication "Economic appraisal of environmental projects and policies. A practical guide."

- *Impact on expenditure patterns*, such as reduced industry, medical and patient costs due to reduced mortality and morbidity. These include medical care (MC) costs saved, avertive expenditures (AE) reduced, and other payments averted (OP). Measurement of these impacts is stated as being important in CEA guidelines, but in general the methods for measurement in health care are not as well developed as in the general (non-health) cost-benefit analysis and development project appraisal literature.

In addition, there are other benefits not related to the health improvement that occur as a result of the environmental improvement (O). These impacts are not often included in CEA, but are usually included in CBA.

Table 4.1. Generic categorisation of benefits to society of health interventions.

Benefit enjoyed by	Type of benefit		Code
Health sector	Reduction in current costs of intervention		**MC**
	Reduction in future costs (less cases, less severe cases)		**MC**
Third party payer	Reduction in pay outs to health care providers[2]		**MC**
Patient[1]	Reduced morbidity and mortality	To be termed: **'health benefits'**[3]	**H**
	Increased life expectancy		**H**
	Increased health-related quality of life		**H**
	Reduced current direct costs of attending care		**MC**
	Reduced future medical or social care costs		**MC**
	Increased productivity		**PR**
	Reduced avertive expenditures		**AE**
Family or carers of patient	Reduced time caring (back to work)		**PR**
	Reduced out-of-pocket payments		**MC**
Industry	Increased competitiveness due to greater efficiency		**O**
	Reduced sick leave of employees (paid sick leave, lost production)		**OP**
	Reduced medical expenses		**MC**
	Reduced damage		**O**
	Reduced avertive expenditures		**AE**
Other government ministries	Reduced running costs or maintenance		**OP**
	Other emergency services (police, fire)		**OP**
	Reduced avertive expenditures		**AE**
All consumers	Reduced running costs or maintenance		**OP**
	Non-health benefits (increased convenience of a good, increased amenity, non-use values (option, existence, bequest) [4])		**O**

TABLE KEY: [1]The patient is the person who *would* have been ill in absence of environmental health intervention. [2] Costs to 3rd party payers are rarely, if ever, included in cost of illness studies. [3] Includes avoidance of grief, anxiety, pain and suffering to the individual, family and friends. [4] Non-use is divided into option value (the possibility that the person may want to use it in the future), existence value (the person values the fact that the environmental good exists, irrespective of use), and bequest value (the person wants future generations to enjoy it). **MC** – medical costs averted; **AE** – avertive expenditure saved; **H** – health benefit; **PR** – production loss saved; **OP** – other payments averted; **O** – other benefits not related to health impact.

Therefore, the primary issue to address is: which benefits should be included in CEA? One of the aims of economic evaluation generally is to identify relevant benefits, and quantify them using appropriate measurement and valuation techniques, with the ultimate aim of maximising societal welfare. *However, which benefits are included and how they are valued depends on the framework of analysis (i.e. CEA or CBA) and the viewpoint of the study.* The main distinction between frameworks is that in cost-effectiveness analysis health benefits are valued in natural (e.g. rates of morbidity) or composite (e.g. DALYs) units, whereas in cost-benefit analysis health benefits are valued in monetary units. The concern of authors in including monetary benefits in CEA is that some benefits to patients are counted twice – i.e. both as the actual health gain, as well as a measure of what individuals are willing to pay for that health gain (Weinstein et al 1996, Leigh et al 1996). On the one hand, WTP of individuals can include all benefits associated with an improvement in health, and thus include health care costs and avertive expenditure saved due to the improvement in health. On the other hand, it has been argued that the willingness to pay for improved health may only contain within it, in addition to the value of reduced suffering, the long term impacts on wealth due to illness, and not short periods of time off work. This argument is made stronger when sick leave is paid by employers (Department of Health, 1999) and when few other expenses are associated with the illness. The UK Department of Health (1999) therefore argues that estimates of WTP to avoid the disutility of ill-health are unlikely to be much "contaminated" by the other costs, such as travel and time costs associated with medical care, medical expenditure and avertive expenditure. This suggests that direct and indirect benefits may be separable, and therefore the former (health gain) is the denominator in the CER, and the latter (money benefits associated with health gain) can safely be subtracted from intervention costs in the numerator of the CER, thus giving a net cost. This assumption has attracted some debate, and Olsen and Richardson (1999) reviewed the state of the debate. A second concern of those conducting CEA of health interventions is that benefits measured that are outside the Ministry of Health perspective will be disregarded, and therefore collecting data on these benefits will not be worthwhile.

While the decision which health related benefits to include in the CER is not free from controversy, the inclusion of benefits not related to health is likely to be considerably more controversial. Current CEA guidelines do not mention benefits unrelated to health. However, *to resolve this question is of central importance for the fair evaluation of environmental health interventions using CEA guidelines.* Some analysts may argue that they should not be included in health care decision making, as the only benefits of interest are health itself or directly related to health. However, if one is taking a societal perspective, it seems wrong not to include all associated benefits, whether related or unrelated to health. A further point is that, if the health sector insists that a health impact assessment is done in all development projects, in order to be consistent economic changes associated with health interventions should also be quantified.

A second important issue concerns whether those willing to pay for a health intervention can be identified so that they can contribute to cost recovery. Table 4.1 shows that there are many different agencies that may potentially be willing to pay for health services that avoid either tangible (health and economic losses suffered) or intangible (emotional impact, non-use value) costs. The aggregation of these tangible costs across agencies (but ensuring that costs are not double counted), gives what is commonly called the "cost of illness" (COI). The aim of the COI approach is to estimate the economic cost of a disease and then to estimate the extent to which a particular intervention will reduce these costs. The COI approach to determining the costs of illness and disease has five components (Hodgson and Meiners, 1979):

1. *Health care direct costs.* These include the medical resources used in treatment and/or prevention (to individuals, household, and government).
2. *Other direct costs.* These include transportation to health care providers, counselling services, and avertive expenditures.

3. *Indirect costs*. These are the costs resulting from losses in economic output.
4. *Social costs and quality of life reductions.*
5. *Overall cost increases throughout the economy.* These include the impact on gross national product when resources are used in health care and thus diverted from other productive uses.

While this classification gives some idea of who would be willing to pay for health improvements, COI studies often do not compare other direct costs (number 2), and they rarely consider the social and overall costs (numbers 4 and 5), or intangible costs, which may be considerable. In practice, COI studies captures the change in economic productivity due to reduced mortality and morbidity, the change in medical care costs, and the change in avertive expenditures. Therefore in considering willingness to pay it is important to include the non-health and non-economic benefits of improved environmental conditions.

Several studies measuring both direct and indirect costs in COI studies have found differing proportions of costs due to these sources. For example, Seethaler (1999) measured only EURO 17.3m for loss of production compared to EURO 147.8m for treatment costs due to morbidity from traffic pollution in France. On the other hand, Leigh et al (1996) found indirect costs to account for a high proportion of total costs of occupational injury in the USA, with US$98bn due to indirect costs and US$50bn due to direct costs. Therefore, the conclusion of these studies is that indirect costs are potentially important.

However, there exist considerable methodological difficulties in the COI approach to indirect cost estimation using the human capital approach[19], and this helped spark interest in CEA as a substitute for CBA (Mills 1985). With the general shift in the health care literature towards CEA as the economic evaluation of choice, the health effect is measured more in terms of change in health status, and not change in production associated with the change in health status. However, more recently, there has been increased interest in incorporating aspects of COI back into CEA, in order to make cost-effectiveness ratios reflect more a societal perspective. As Mills (1991) states, this methodological refinement has tended to blur the distinction between CEA and CBA, especially when benefits in the latter are based on the human capital approach.

4.3 Environmental health interventions

4.3.1 Water, hygiene and sanitation

The problem of deciding which benefits to include in WHS interventions has been a subject of discussion of many of those working in this field. Quoting examples from Berman (1982) and Briscoe (1984), Feachem (1986) writes that "…special difficulties are inherent in applying CEA to interventions having multiple benefits, and WHS present these difficulties in an extreme form. In addition to their impact on diarrhoea rates among young children, these interventions may avert diarrhoea in other age

[19] Not only do indirect costs saved rely on uncertain estimates of the value of health using the human capital approach, but also COI is not relevant for assessing efficiency in the use of resources at the margin. On the other hand, COI studies help raise the profile of important diseases with a high epidemiological and economic burden. Data from COI studies can also provide useful baseline data for epidemiological studies and economic evaluations.

groups, reduce the incidence of other infectious diseases and have a variety of benefits unrelated to health". Philips (1993) also argues that if optimal resource allocation is to be achieved, it is important that all ramifications of an intervention should be taken into account. For example, curative medical interventions are targeted mainly at diarrhoea, whereas preventive WHS interventions have a broader range of health effects, such as reduced skin disease, hepatitis A, and intestinal worms. Therefore, the results of Varley (1998), who modelled the cost-effectiveness of WHS interventions on diarrhoeaa incidence in under 5s, would have underestimated the overall health benefits and thus the true cost-effectiveness of WHS interventions. In conclusion, it is important for the analyst of WHS interventions to adequately represent the diverse range of health benefits that they provide in CEA. In defining COI contents, PEEM (1993) includes:

1. Savings in costs of curative and preventive care (reduction in disease cases that would have been treated × costs of treatment per case).
2. Gains in production of cases averted (work days increased × value of average day not worked).
3. Gains in production of deaths averted (work years increased × discounted value of average income per year).

However, this discussion is related purely to health benefits. WASH (1991) recommends casting a wider net still in measuring the benefits of WHS interventions. They state "benefits analysis related to water supply and sanitation projects should include measurement of direct economic benefits, such as increased time availability when water is more conveniently located, commercial benefits (reflected in infrastructure improvement leading to increased investment and other opportunities) and health benefits, both direct in terms of avoided medical expenses[20] and indirect in terms of productivity gains due to reduced morbidity".

Table 4.2 below summarises the main benefits of WHS interventions. No studies categorised health benefits according to the PEEM (1993) classification, nor measured proportional gains from each benefit source. It is clear from the table that there are likely to be significant indirect benefits related to health, as well as benefits unrelated to health. For example, willingness-to-pay studies have shown that reduced collection time has considerable value to households and the individuals within them, which is critical in the light of the burden placed on women in poor societies (Briscoe, 1984). A study by Caruthers (1973) found that 100 minutes per household per day were saved as a result of improved water supplies in Kenya. Whittington and others have shown that even poor people are willing to pay significant amounts for improved water supply. However, these studies rarely separate health from non-health benefits. In addition to time saved, a reliable water supply allows further possibilities to run a business from home or buy labour saving devices. Also, risk aversion measures such as boiling water do not need to have large amounts of money or time spent on them. Finally, WTP studies on the amenity benefits of ground water have indicated that individuals are willing to pay significant amounts of money in developed countries for the amenity benefits such as recreational use and non-use benefits.

[20] It should be noted, however, that decreased mortality could result in higher direct costs if the lower mortality is associated with increases in long-term morbidity.

Table 4.2. Categorisation of benefits to society of water and sanitation interventions.

Benefit enjoyed by	Type of benefit	Code
Health sector	Reduction in current costs due to health intervention: materials such as oral rehydration therapy and antibiotics, staff time	MC
	Savings in poison control centre costs	MC
	Reduction in future costs (less cases, less severe cases)	MC
Third party payer	Reduction in pay outs to health care providers	MC
Patient[1]	Reduced morbidity and mortality	H
	Increased life expectancy	H
	Increased health-related quality of life	H
	Reduced direct costs of attending care (out-of-pocket expenses)	MC
	Reduced future medical or social care costs	MC
	Increased productivity or capital formation activities such as less time off work and school and increased efficiency while at work or school	PR
	Reduced risk avertive expenditures such as money cost and time input	AE
Family or carers of patient	Reduced time caring (back to work)	PR
	Reduced out-of-pocket payments for medical care	MC
	Reduced risk avertive expenditures (see above)	AE
Industry	Direct economic value of high quality water such as irrigation water for crops, fishery production, and sea ecosystems	O
	Reduced sick leave of employees (paid sick leave, lost production)	OP
	Reduced medical expenses	MC
	Reduced avertive expenditures	AE
Other government ministries	Reduced running costs or maintenance	OP
	Reduced avertive expenditures	AE
All consumers	Reduced running costs or maintenance	OP
	Non-health benefits such as increased convenience of a water supply, increased amenity (laundry, recreation), and non-use values (option, existence, bequest)	O

TABLE KEY: [1] The patient is the person who would have been ill in absence of environmental health intervention. *Abbreviations:* MC – medical costs averted; AE – avertive expenditure saved; H – health benefit; PR – production loss saved; OP – other payments averted; O – other benefits not related to health impact.

An example of the importance of averting the indirect economic impacts that result from poor water supply and sanitation is available from a study by Paul and Mauskopf (1993) which studied the impact of the cholera epidemic in Peru (January 1991 to March 1992). In this epidemic, Paul and Mauskopf (1993) estimated that three-quarters of the economic costs were from the indirect productivity losses due to morbidity (US$2.6m) and mortality (US$93.9m), as well as the macroeconomic impact of loss of exports (US$8.1m), tourism (US$15.4m) and domestic production (US$26.9m). Out of a total economic loss of US$200m, it was estimated that only US$53m was met by the health sector in terms of treatment of cholera cases and public education campaigns. Therefore, these data suggest not only that other costs should be measured to show the full picture, but also that other beneficiaries, such as consumers, industry and other government departments, would be willing to pay to prevent such an outbreak (and the associated sequelae and secondary outbreaks) from happening again. WASH (1991) also recognise longer term impacts of WHS interventions, such as changes in population pressures through decreased mortality, or changes in physical capital formation through savings rates and school attendance. However, inclusion of all these benefits represents a considerable challenge for the analyst.

In addition to initiatives taken by the health sector to improve or sustain the environment, the potential costs of adverse environmental impacts of industrial development need to be identified. This is essential to identify the true cost-benefit of a development project, by taking into account adverse health effects, but also it identifies costs that should be paid by industry for any preventive measures or towards the costs of cleaning up adverse impacts. In the case of avoiding adverse impacts on water quality, human health is not the only benefit, as there may be other losses avoided, such as agricultural productivity loss (less irrigation water and polluted waters), fish productivity, amenities (laundry, bathing), and sea and freshwater ecosystems (Dixon et al 1986). Further questions concern whether or not the Polluter Pays Principle can be extended to health impacts through less complicated mechanisms than legal action, and whether payments or fines flow back to the health sector.

4.3.2 Food safety

Buzby and Roberts (1997) list different sources of the societal costs of foodborne illness, shown in Table 4.3. Costs of illness are borne by humans, industry and the regulatory body. Under human costs avoided are medical costs, income or productivity loss, other illness costs, psychological costs, and averting behaviour costs. Industry costs avoided consist of the costs of animal production due to disease (including costs to workers of illness), and outbreak costs such as herd slaughter and reduced product demand. Under regulatory body costs avoided are the additional costs of outbreak and cleanup. Therefore, there are potentially considerable health and non-health benefits of avoiding an outbreak of foodborne illness, and most of these accrue outside the health sector[21].

From experience in developed countries, the beneficiaries of improved food safety may be willing to pay for the cost of regulations, not only because it is good business practice, but also it avoids putting consumers at risk with associated costs of illness. Obviously the amount beneficiaries are willing to pay for a particular food safety measure depends on the severity of the repercussions (such as law suits against companies, or long periods out of work for the victims). However, health care decision makers are unlikely to include the benefit to industry or food price increases.

Table 4.3. Benefits related to food safety interventions (cost of illness).

Human benefits	Industry benefits	Regulatory body benefits
Reduced morbidity and mortality Medical costs of illness[1] avoided • Health sector • Patient Income or productivity loss Avoided costs • Travel costs for treatment • Child care costs • Lost leisure time • Psychological costs • Averting behaviour costs[2] Altruism (WTP for others to avoid illness)	Costs avoided due to avoiding outbreak of illness • Herd slaughter / meat spoilage • Disposal of contaminated animals • Plant closings and cleanup • Worker illness from animals • Regulatory fines • Liability suits from customers • Reduced product demand • Advertising following outbreak	Costs avoided due to avoiding outbreak of illness • Costs of investigating outbreak • Costs of cleanup • Increased surveillance following outbreak

(Taken from Buzby and Roberts 1997, pages 58-60).
TABLE KEY: [1] These include physician, laboratory, drug, emergency room, poison control centre, hospitalisation, and other institutional care costs. [2] Includes extra cleaning/cooking time, extra costs of refridgeration, flavour changes, and increased prices of safe food.

[21] However, ultimately the cost of improvements in food are met by consumers, either in the form of direct expenditure on more expensive food items, or indirectly through the taxation system.

In addition to preventing foodborne illness, there are also benefits associated with *post-event* interventions in foodborne illnesses (therefore they may not fall under 'environmental' health intervention). Several studies conducted in the USA on the WTP for poison control centres showed not only that individuals are willing to pay significant amounts of money for poison control centres, but also that the cost savings to emergency and inpatient departments of hospitals due to poison control centres are considerable. For example, Miller and Lestina (1997) estimated that for every US$1 spent on poison control centres in the USA, US$6.50 was saved in medical care payments. People have been found to be willing to pay for the increased convenience of poison control centres (and an emergency hotline) and the increased expertise they provide. However, no studies separated the health and non-health benefits in the WTP estimates.

4.3.3 Vector control

All vector control measures have significant associated economic benefits resulting directly from the reduction in ill-health (see Table 4.4). Vector control has received considerable attention from economists recently, with several CEA, WTP and productivity studies.

The range of options for vector control (chemical, biological, environmental management and personal protection measures) shows significant variation in the scope of benefit patterns they represent. The application of pesticides for vector control, be it spraying of the internal walls of houses with residual insecticide, fogging of insecticide in the case of epidemic outbreaks of a vector-borne disease or the application of larvicides to specific breeding sites, is most confined in its benefits. Beyond the immediate goal of reducing disease transmission risks (sometimes for multiple diseases: in the past, spaying for malaria vector control has had a significant impact on filariasis and/or leishmaniasis transmission) there are currently no other benefits. The intense spraying programmes under the malaria eradication strategy had considerable social benefits in terms of employment generation. Pesticide production also supports an entire production sector in the world economy, but the market share of public health pesticides is an order of magnitude below that of agricultural pesticides and is shrinking.

Biological control can have non-health benefits in some cases, such as the introduction of larvivorous fish. The sustainability of such programmes, particularly when they are community-based, often depends critically on the non-health economic benefits, in this example on the promotion of joint cultivation of larvivorous fish and fish with a nutritional and therefore economic value.

Environmental management shows enormous variation in different eco-epidemiological settings, and non-health benefits vary accordingly from one location to another. The health benefits of both biological control and environmental management are limited by the fact that they reduce, in first instance, vector population densities and, unlike residual pesticides, do not affect vector longevity. There is also great variation in the scale of the non-health benefits: reservoir management for vector control may, downstream from a large dam, provide major environmental benefits in terms of wetlands protection or maintenance of traditional agriculture. Small-scale operations, such as the collection of algae or aquatic weeds from ponds or tree planting to dry out marshy areas, may be the basis for community based cottage industries. Infrastructural improvements may have primary health objectives (e.g. the use of automatic syphons to flush streams to reduce mosquito breeding in Malaysia), but more generally larger infrastructural works (road, dams, irrigation schemes) should undergo an impact assessment to identify opportunities to incorporate health safeguards in their design and operation.

Personal protection, by its very nature, implies individual or household health benefits, but the scaling up of, for example, the use of insecticide impregnated mosquito nets and other impregnated textiles, has the potential of creating opportunities for local industry.

Table 4.4. Health and non-health benefits of vector control interventions

Benefits related to health effects	Benefits unrelated to health effects
Reduced mortality	Secondary sources of income
Reduced morbidity	Public good benefits such as environmental protection
Increased quality of life and life expectancy	
Reduced curative and preventive care costs	Improved quality of life (nuisance mosquitoes)

Many of the benefits listed in Table 4.4 are not easy to measure and questions about valuation methods raise even more fundamental questions about whether all hypothesised benefits should be included at all. For example, Mills(1991) notes that to measure production losses for all those affected using an average wage would probably overestimate the actual economic losses. First, not all those disabled would have been productively employed in market or non-market production, especially at slack times of the year. Second, studies have shown that production loss may be reduced by substitute workers. These problems are tackled in the valuation section later, but the relative size of this benefits has implications for the importance of including them.

4.3.4 Waste management

There are also a wide range of additional benefits associated with the appropriate management of waste, whether household solid waste, medical waste or chemical waste from industry. Section 4.2 listed Postle’s (1997) categorisation of benefits to individuals, to production and consumption, to economic assets, and to environmental assets. Table 4.5 shows that most of these benefits are actually unrelated to health, and include amenity value of cleaner neighbourhoods, reduced costs of alternative means of waste disposal, and production/consumption benefits. Some of these benefits occur through averting the release of chemicals into water courses (thus causing cases of water and food poisoning) and into the air (e.g. causing acid rain, and the effects on agriculture and materials) and are dealt with under the WHS and air sections, respectively. However, some benefits are very indirect, and may take some time to be felt.

Waste management is potentially not a very cost-effective intervention using health benefits alone, but it does have significant amenity and non-use values. Disamenity of landfill sites was studied by Brisson and Pearce (1993). Agencies potentially willing to pay include those that generate waste (general consumers, businesses and industry) as well as local councils, although payments are unlikely to be voluntary in nature. Also, because the health effects are felt by the poorest members of society (those scavenging waste dumps), it may receive support from central government. However, there may be income losses from the informal sector (those who collect and resell waste), which also need to be estimated.

Table 4.5. Health and non-health benefits of solid waste disposal.

Benefits related to health effects	Benefits unrelated to health effects
Direct health impacts[1]	Costs saved from electricity from incineration
Reduced treatment for illness	Reduced private costs of solid waste disposal
Reduced avertive costs	• Individuals
Increased productivity due to cases and deaths averted	• Companies
	Production and consumption
	• Crops, forests, fisheries
	• Water-using industry
	Economic assets
	• Materials (corrosion, soiling)
	• Property values
	Environmental assets
	• Amenity value (e.g. recreation)
	• Non-use

TABLE KEY: [1] Due to reduced contact of vulnerable populations with rubbish in street, rubbish dumped illegally, and improved management of official land fill sites.

4.3.5 Air pollution

The benefits from air pollution reduction include both health and non-health impacts. Many studies have estimated dose-response function (DRF), particularly for semi-particulate matter, and the associated direct and indirect costs. The DRF is used to estimate the number of cases of deaths, respiratory infections and other health impacts, and thus restricted activity days or work loss days. Benefits unrelated to health effects of improved air quality are likely to be smaller in proportion, such as greater amenity due to visibility and recreational activities, and less damage caused by acid rain. The UK Department of Health (1999) identified a number of different aspects of air pollution induced ill-health: private costs of dealing with illness, public costs of treating illness, loss of output, and wider disutility costs of ill-health. Department of Health (1999) measured the financial costs of dealing with the adverse effects of air pollution on health and how, if air pollution were reduced, such expenditures would be affected. Included were costs falling on national health service budgets (inpatient and outpatient visits and the use of pharmaceuticals) as well as costs to the patients themselves, and loss of production due to time spent ill. However, production consequences, amenity and non-use values were not considered. In addition, certain gases ('greenhouse' gases) that cause air pollution are also primarily responsible for climate change.

Table 4.6. Health and non-health benefits of reducing air pollution illness.

Benefits related to health effects	Benefits unrelated to health effects
Direct health benefits (acute, chronic) Medical costs averted (current and future) • Emergency room visits • Primary health care (e.g. asthma cases) • Hospitalisations Out of pocket costs associated with medical care avoided Economic productivity due to improved health and LE • Restricted activity days or work loss days Time receiving medical care Avertive costs (e.g. cycling mask, moving home or job) to avoid effects of air pollution	Amenity • Visibility • Recreational activities Value of economic assets (residential, commercial, government) • Materials (corrosion, soiling) • Property values Overall climate effects Non-use benefits Impact on environment (use and non-use) • Agriculture • Forestry • Fishing • Ecosystem Indirect effects of less traffic • Less noise • Less accidents • Alternative land use • Property prices

(assembled from various sources, including Bartonova (1997) and Seethaler (1999))

While agents may be willing to pay for improved air quality, the public good nature of air makes it very difficult to charge according to actual consumption. On the other hand, individuals affected (e.g. those within city limits) could be taxed to pay for pollution reduction measures. Alternatively, the polluters themselves could be charged for the pollution they produce, and thus encourage them either to reduce pollution emissions or move out of suburban areas. Charging the polluter directly is easier than charging the beneficiaries.

4.3.6 Climate change and stratospheric ozone depletion

The health and non-health benefits of slowing, stopping or reversing global climate change and stratospheric ozone depletion are highly uncertain, due to uncertainties about the rates of change, and whether preventive measures actually have an impact in the short term (some may be irreversible in the short term – e.g. refreezing polar ice caps). However, impacts are hypothesised through likely changes in global temperatures and seasonal patterns, rising sea levels, exposure to ultraviolet rays, and worsening air quality (WHO 1997). Both direct and indirect costs are likely to be significant, caused by thermal stress, cancers, disease vectors, etc. Averted expenditures and economic losses saved due to avoiding ill health are expected to be enormous. In addition to the economic losses associated with health impacts, other economic losses are very great indeed, such as forced migration due to sea level rise, impact on production activities, and food insecurity. In addition, as climate change involves the very future of the planet, non-use values are likely to be very significant. However, there is considerable uncertainty associated with both cost and health impact estimation, as the future impacts of pollution and pollution abatement measures are themselves extremely uncertain. Uncertainty is dealt with further in chapter 8.

Table 4.7. Health and non-health benefits related to stopping or reversal of climate change and stratospheric ozone depletion.

Benefits related to health effects	Benefits unrelated to health effects
Reduced mortality and morbidity due to • Thermal stress • Malaria • Drowning • Cancer (ultraviolet rays)	Food production and insecurity Migration of populations away from danger areas Psychological stress of resulting insecurities Long term economic productivity
Medical expenses Economic productivity Avertive expenditures Long term quality of life of world population	Non-use values • Existence • Bequest

The willingness-to-pay to avoid the adverse effects of global warming is similar as for air pollution, but more difficult to measure exactly. Obviously many developing countries are powerless to reverse climate change, other than to adopt non-polluting technologies, which are still relatively expensive. Non-use values could be elicited, and money extracted using voluntary or mandatory taxes. However, the costs and benefits are much wider than the health sector would wish to consider, although it should play a central role in mitigating the health impacts.

4.3.7 Occupational health and safety

The topic of the economics of occupational health and safety has received much attention from economists both in government and industry, as described in Chapter 3. Although costs are often considerable, and regulations mandatory, the resulting benefits through a healthier workforce and increased productivity are potentially large enough to more than pay for the costs. Most studies focus on the benefits related to improved health.

Benefit studies such as cost of illness were of three main types – national, company and individual. Most studies were done at the company level. These studies measured a range of benefits in estimating cost-benefit ratios or the costs of illness. A range of benefits of occupational safety and health interventions are presented in Table 4.8. Studies have shown that the benefits included have important implications for conclusions. For example, Fahs et al (1997) argues that econometric models to estimate net social benefits of occupational risk reduction significantly underestimate the net social benefits due to omitting the economic benefits of healthier workers. Conversely, they argue that the models tend to overestimate the social costs of industry regulation, since potential savings in averted health care costs are ignored for worker whose occupational disease is prevented. Another choice for benefit inclusion concerns which occupationally related injuries and diseases to include, since some will account for the majority of costs. For example, Fahs et al (1997) focus their attention on the six leading causes of death from occupationally related illness in the US in 1992[22].

[22] These are cancer, nervous system disorders, pneumoconioses, chronic respiratory disease, cardiovascular and cerebrovascular disease, and renal disorders.

Table 4.8. Health and non-health benefits of occupational safety.

Benefits related to health effects	Benefits unrelated to health effects
Individuals	**Individuals**
Health benefits	Improved job satisfaction
Income losses & fringe benefits averted from reduced illness or injury	Improved working climate
Medical care costs avoided	
Non-medical rehabilitation costs	
Unpaid family or community care	
Reduced home production and leisure time of worker	
Reduced avertive expenditures due to safer environment	
Company	**Company**
Medical care costs avoided	Productivity of new machinery
Non-medical rehabilitation costs	Improved quality of products
Absenteeism or sick leave (productivity impact)	Competitiveness of regulated industry
Personnel turnover costs and friction costs	Improved working climate
Administration costs of sickness absence	Less damaged equipment
Reduced liabilities and legal costs	Company image effects
Reduced danger money in wages	Impact on non-economic company values
Reduced avertive expenditures due to safer environment	
Reduced social security disability benefits	
National	
Lost production	
Morbidity (days spent not producing/earning)	
Mortality (loss of production)	
Police and fire services not used	

(taken largely from Mossink et al, 1998)

Production losses, medical costs and avertive costs saved as a result of improved occupational safety have been shown to be enormous, even for developed countries that already have strict occupational safety regulations. Mossink et al (1998) emphasise the need to take a societal perspective, as investments at the company level eventually means less expenditure by government on health care and less expenditure by individuals on avertive or corrective behaviour (Aaltonen and Soderqvist, 1988). Also, benefits that result from improved health are significant and the argument for their inclusion in economic evaluation is strong. For example, Leigh et al (1996) estimate that the direct costs (medical, administrative, property damages, police and fire services) constitute 33% of total costs, and indirect costs (lost wages, fringe benefit loss, loss of home production, hiring and training costs, time delays, and family home health costs) 67% of total costs of occupationally-related injury in the USA[23]. Occupational disease cost US$25.5 billion in the Leigh et al study, fatalities making 77% of the cost. The main types of fatal disease were cancer (US$9.4 billion), cadiovascular and cerebrovascular disease (US$5.8 billion) and coronary respiratory disease (US$3.9 billion).

Mossink et al (1998) tabulated the benefits included in a chosen sample of economic OSH studies from Europe, with medical care costs and labour market costs (sickness absence, permanent disability, death) as the major components of cost of illness studies. Other components of cost-of-illness, such as national and company level costs of administration, damaged equipment and police costs, were given little attention.

[23] These figures are close to the 1992 estimate made by the National Safety Council of US$116 billion, with US$62.5 for wage and productivity losses and US$22 for medical expenses.

Davies and Teasdale (1994) estimated both financial losses as well as quality of life changes to individuals as a result of work related accidents and ill health. Financial losses included both lost income and expenditure on drugs and hospital attendance. However, they prefer the use of WTP (contingent valuation) as opposed to the human capital approach (potential future earnings) (see Chapter 6 for explanations of these approaches). However, if there is a social security system in operation, individuals will not suffer such great losses.

The valuation method chosen has been shown to be important in measuring potential benefits (covered more in Chapter 6). According to Leigh et al (1996), WTP to avoid injuries, illness and death in the work place is in fact far greater than was valued using the human capital approach. Using labour market data, value of life estimates range from US$3m to US$16m in 1990 prices. Applying the value of a saved life to the number of deaths, US$27 billion was due to injury deaths, and US$532 billion was due to injuries. Disease deaths amounted to US$242 billion and non-fatal disease to US$34 billion. There were no WTP estimates available for occupational disease, while they were available for injuries. These figures are considerably more than the Fahs et al (1997) figure of US$25.5bn (1992 dollars), but this is partially explained by the consideration of only 6 diseases.

Non-health benefits have been given little attention, although several studies have recognised increased productivity due to rationalisation associated with OSH interventions, as well as less damaged equipment and buildings through less accidents and fires. For the individual, there may be greater job satisfaction and improved working climate, but no studies have examined these benefits.

In terms of willingness to pay for implementation of OSH measures, Mossink et al (1998) discuss the importance of cost internalisation, whereby companies take full responsibility for accidents and diseases at work. This makes them more likely to be motivated to take preventive measures. While Mossink et al (1998) doubt whether full cost internalisation can be achieved, they outline a number of options for cost internalisation, including:

1. Differentiation of insurance premiums for companies according to safety and health risks or numbers of accidents and diseases in the past.
2. The right of workers to claim compensation for the costs related to occupational accidents and diseases.
3. Limiting the possibilities of companies insuring the costs of sick leave.
4. Full cost pricing – where products sold on the market place include the costs for OSH investments and damages due to work-related illnesses.
5. Increase availability of cost effectiveness studies to point out the pay back potential of OSH investments.

4.4 Discussion and conclusion

The key issues for discussion include the likely importance of non-health benefits compared to health benefits, whether they should be included in CEA, and how beneficiaries can be made to pay for the benefits.

There are clearly multiple non-health benefits from environmental health interventions, accruing to a variety of beneficiaries. For most environmental health interventions there are considerable economic gains unrelated to health due to environment improvements, such as greater productivity and reduced damage, as well as intangible benefits such as psychological, amenity and non-use benefits. Non-use benefits are divided into: (a) option value - the possibility that the person may want to use it in the future; (b) existence value - the person values the fact that the environmental good exists, irrespective of use; and (c) bequest value - the person wants future generations to enjoy it. However, current empirical evidence is poor for intangible benefits, and therefore the relative proportions of these and tangible health and non-health benefits are unknown. Non-health benefits are expected to vary between different environmental health interventions. One study found that non-use benefits constituted 40% of total benefits of preserving woodland in Colorado, although this was not a health intervention. However, as non-health benefits are important for environmental health interventions compared to core or curative health services, it is important to include them to improve their relative cost-effectiveness. While current CEA guidelines recommend a societal perspective, they do not specify whether or not non-health benefits are relevant to the CEA. No mention is made of non-health related production, amenity and non-use benefits for the simple reason that these benefits rarely arise from core health services.

However, conceptually the benefits side of the equation should include the willingness to pay of beneficiaries, and if they are willing to pay for non-health benefits, then it should be included. The WTP method is useful in valuing non-health benefits for two reasons:

1 It gives a monetary value for an improved environment, and
2 It suggests to decision makers ways to recover costs based on a hypothetical market situation.

If costs can be recovered from the environmental health intervention, then the health service is more capable of taking a societal perspective, one that include non-health benefits. This has clearly worked in some areas of environmental health, such as water supply and sanitation, where customers are charged fees, and the health sector only pays for the contribution it makes in its own area of expertise. However, the potential for cost recovery depends on the nature of the good, whether public (and therefore consumption non-excludable) or private, or somewhere in between (for example, a charging mechanism is used but does not charge according to use).

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A possible problem, as noted above already, is that while WTP can be designed to be comprehensive, incorporating all aspects of an individuals' welfare function[24], it cannot be used in CEA when it means double counting the benefits of a health intervention. However, if the health aspect of the WTP can be taken out (see footnote 24) then there should be less objections to its use in the cost-effectiveness equation. Also, WTP of all agents who stand to benefit from the intervention can be taken into account (see Table 4.1).

While inclusion of these other benefits is justified under the adoption of a societal perspective, such a perspective is hampered by various imperfections in the decision making process. One significant barrier to adopting a societal perspective in economic

[24] It should include the WTP of the individual for the health improvement, not only health per se, but also the reduced medical costs, time costs, productivity loss, and avertive expenditures.

evaluation is the cross-agency nature of environmental health interventions, leading to communication problems, inadequate data, and limited levels of 'joint' decision making between ministries that are in effect maximising different objectives. Possible ways in which the health sector can be empowered include:

1. Adjusting the remit and strengthening environmental health departments, and the structure in which they operate within the ministry of health.
2. A proper legal framework within which regulations can be developed and enforced (to ensure the latter, any regulatory component in an environmental health law should come up with a budget approximation).
3. A policy framework conducive to inter-sectoral planning and decision making.

Another problem is that of cost recovery when not all the benefits are realised instantaneously, whereas significant costs may need to be recovered in short-term. This budget constraint means that while many agents may show high WTP for an environmental improvement to occur in hypothetical surveys, few can provide the funds that represent the benefits they will enjoy over the life-time of the environmental improvement. Also, the uncertainty inherent in environmental improvements and its impact will further reduce the willingness of beneficiaries to part with their money. Finally, it is possible that benefits are perceived differently by those benefiting compared with the public agency making the evaluation.

However, it could be argued that, just because indirect effects exist, they should not necessarily be included in the cost-effectiveness ratio. The CER can only realistically incorporate all benefits due to an intervention when the decision maker can take the perspective of all public sector agencies combined, such as the Ministry of Finance, but not one ministry in isolation. While on the one hand the inclusion of 'other benefits' may increase the relative cost-effectiveness of environmental health interventions, their inclusion means that their full costs should also be included (see Chapter 5). As discussed above, a considerable amount of the costs may have to be recovered outside the health ministry. The issue of cost recovery is partly addressed by identifying what the key inputs are to environmental health interventions, who is responsible for them, and therefore whose collaboration and funding is essential to the success of the intervention. For example, several government departments may have to collaborate to provide a single service, or private enterprises may be key players in following regulatory advise, and thus in spending the necessary resources on improvements. This reliance on other agents can also hinder decision making, as they may not have the funds or consider the intervention a priority. On the other hand, a regulatory framework is a useful way of motivating some agents, particularly in the private sector, with the threat of fines or licenses being taken away.

5. COST INCLUSION

5.1 General issues

This section reviews which costs of environmental health interventions could or should be included in cost-effectiveness analyses under a variety of perspectives. First the full range of possible costs of environmental health interventions is listed. In particular, the advice of current CEA guidelines with respect to the inclusion of costs is compared with those costs identified in the environmental health literature. Later sub-sections review costs specific to each major area of environmental health intervention in turn. In so doing, several issues are addressed:

- What is the possible impact of the inclusion/exclusion of non-health sector costs on the CEA results? What proportion of costs fall within and outside the health sector?
- To what extent have previous studies measured costs comprehensively?
- To what extent should the health sector and different decision makers within it be interested in non-health sector costs?
- Does the identification of costs suggest means of cost recovery for environmental health interventions? Should costs be paid by those whom they fall on, or should subsidies be used? (although this point is not related directly to the CER, if costs can be recovered partially or fully the decision maker may be more interested in including them in the CER)
- The role of environmental health departments within the Ministry of Health – whether they play a marginal role in formulating health policy, or whether they can provide a critical link between the Ministry of Health and others sectors – will to a large extent determine which costs and benefits the Ministry of Health is likely to find acceptable to include in cost-effectiveness analyses.

As in the case of benefit inclusion, there are several different categories of cost in implementing environmental health interventions, and some of these may not be borne by the health sector or patients directly affected by those whose health is directly at risk from the environmental hazard in question. As mentioned in the previous discussion, the collaboration and funding of other agents is often essential in improving the environment, and it is essential to motivate these other agents in order to secure the health gains. However, the question arises concerning which costs should be included in the CER when not all costs are met by the health ministry.

Therefore, it is necessary to make a full elaboration of costs of health interventions, before discussing them in the context of specific environmental health interventions. Table 5.1 below shows that most of the costs of environmental health interventions do not fall on the health sector or patient, and they involve very limited use of health services. This occurs because environmental health interventions are generally preventive in nature, and therefore they do not involve treatment of specific individuals, but the lowering of the risk of unaffected individuals whose likelihood of illness may or may not be known in advance (therefore they are called 'consumers' instead of patients). Therefore, many of the costs fall on industry, consumers and other government ministries or local councils (depending on responsibility). This makes environmental health interventions clearly different from most health services, where a narrow perspective would not capture the costs of other agents. From Table

5.1, it is clear that costs fall on a much broader group of agents for environmental health interventions than for most health services.

5.2 Environmental health interventions

Table 5.1 indicates which agents bear the costs of the different environmental health interventions. The following sections discuss the costs of each intervention in greater detail, describes what cost estimates have been made in the literature, and the implications of cost inclusion in the CER.

5.2.1 Water, hygiene and sanitation

The costs of WHS interventions have been presented very little in the health literature, and usually crude approximations are made instead of detailed estimates. Table 5.2 shows that a wide range of costs fall on agents outside the health sector. For example, Varley et al (1998) estimate that 'hardware costs' cost US$72 per household per year, compared to 'software costs' of US$3 per household per year. This means that a high proportion of WHS costs, the hardware costs, are traditionally not paid for by the health sector. Who meets these hardware costs varies between location, and may be raised in part or full from charges to users, community funds, or government subsidies (WHO 1994) and will depend on setting.

Table 5.1. Generic categorisation of costs of health interventions, by type of intervention.

Cost borne by	Type of cost	Intervention
Health sector	**Categories**	ALL
	Costs related to intervention (normal life span)	ALL
	Costs related to intervention (extended life span)	ALL
	Costs unrelated to intervention (normal life span)	ALL
	Costs unrelated to intervention (extended life span)	ALL
	Non-health care costs in extended life span	ALL
	Types of care	
	Inpatient, outpatient, community care	CORE
	Outreach doctor or outpatient care	WORK
	Health education outreach and media	WHS, WAST, FOOD
	Poison control centres	WHS, FOOD
Other	**Types or sources of cost**	
Third party payer	Costs of care to insured individuals	CORE
Patient	Insurance costs	ALL
	Out-of-pocket expenses	CORE
	Lost income during medical care	CORE
	Time expenses for travel to medical care	CORE
Family, carer or community	Accompanying patient	CORE
	Lost income or education opportunities	CORE
	Time expenses for care	CORE
	Out-of-pocket expenses	CORE
Industry[25]	Compliance with emissions regulations	WHS, AIR, CLIM
	Compliance with waste disposal regulations	WAST
	Compliance with work safety regulations	WORK
	Compliance with food safety regulations	FOOD
	Compliance with home fuel production regulations	AIR
	Compliance with drinking water regulations	WHS
Agriculture	Change in land use following water management	VECT, WHS
Local council	Waste disposal services	WAST
	Water clearance activities	VECT
	Local management of water supply and safety	WHS
	Traffic control measures	AIR, CLIM, OSH
Other government	Check compliance with regulations	ALL
	Providing clean water	WHS
	Laying water and sewerage pipes	WHS
	Water clearance activities	VECT
	Education activities	ALL
Consumers	Compliance with emissions regulations	AIR, CLIM
	Change in transport patterns	AIR, CLIM
	Compliance with waste disposal regulations	WAST
	Compliance with food safety advice	FOOD
	Compliance with home fuel advice	AIR
	Compliance with water clearance advice	VECT
	Increased prices passed on by industry	ALL

TABLE KEY: ALL – all interventions; CORE – core health services only; WHS – water, hygiene and sanitation interventions; FOOD - food safety; VECT – environmental vector control interventions; AIR – air pollution interventions; WORK – occupational safety; CLIM – climate change interventions; WAST – solid waste disposal.

[25] As already discussed, environmental health interventions often come in the form of regulations, such as measures to reduce air pollution, or OSH interventions. Regulations that impose a net cost can have two principal types of effect. The first is when a regulation applies at the local level only. In this case it imposes costs on a producer and makes it less competitive. Under perfect competition the producer cannot increase prices, and so it either earns less profit, pays lower wages, or goes out of business. The second is when a regulation applies at the entire industry level. In this case costs are passed onto consumers in a higher price, or companies attempt to cut costs to keep prices stable. These points should be borne in mind when examining the impacts of regulations on private enterprises.

There is no general agreement concerning which costs should be included in the CER. Briscoe (1984) suggests including the costs of WHS activities minus the amount that users are willing to pay (therefore the net cost to the providing agency). On the other hand, Varley (1998) recommends focussing on the cost of WHS interventions to the health programme budget. This latter approach is justified in that it yields results that are more informative and useful for allocating health programme resources, and results such as those of Varley et al (1998) show WHS interventions to be comparable to oral rehydration therapy in terms of cost-effectiveness, especially when hardware already exists and the intervention involves only adding software. However, their method implicitly assumes a zero opportunity cost for non-health programme resources used for WHS interventions, and thus is unlikely to make optimal use of society's resources. For this reason, WASH (1991) state that "comprehensive analysis of the economic effects of water supply and sanitation services have to include cost analysis components, such as construction costs, costs related to community organisation and participation, training, and ongoing operations and maintenance".

Table 5.2. Likely health sector and non-health sector WHS intervention costs

Health sector costs	Costs outside health sector
Planning WHS interventions cost Software costs • Education • Social marketing • Research costs • Monitoring and surveillance	Hardware costs • Pipes • Latrines • Digging equipment and materials • Staff time • Repairs Water quality maintenance • Treatment • Finding new water sources User costs (fees)

WHO (1994) produced a booklet outlining issues in financial management of water supply and sanitation. They identify the costs of a sustainable WHS system as the sum of labour, goods and services, and capital resources mobilised. Activities include: preliminary investigations, construction or rehabilitation, major repairs, operation, maintenance, and management and overheads. They encourage collection of information on all areas where costs will arise, including making projections of demand, estimates of water losses, seasonality in supply and demand, and requirements for wastewater collection. WHO (1994) also discusses cost recovery (initial, recurrent) options, which include user charges (connection, fixed, metered), water vending options (when not all households served by pipes), community fund raising (e.g. revolving funds), grants from external and local funding institutions (local indirect taxes), loans, and funds from government. They recognise that while health care initiatives and their implications for WHS planning are important, other agencies are interested in the uses of water (agriculture, industry), and therefore may want to be involved from early stages, which may involve contribution to costs. Johansson (1987) provide pollution abatement costs, but little evidence is available of other types of WHS interventions.

5.3.2 Food safety

Most of the costs of food safety measures are likely to fall outside the health sector, as changes in production/processing will require the bulk of costs. However, there are also costs associated with planning and monitoring safety regulations, which may be partially met by the food industry, but also government departments including the health ministry. For those costs met by private enterprise, it is likely that eventually the consumer will pay in the form of higher prices. Consumers may also be involved in preventive measures, such as the purchase of refridgerators or containers for safe storage, or alternative eating habits. While it is essential to include all relevant costs associated with food safety measures, it is important to avoid double counting. For example, producers may pay government for monitoring costs, therefore the charges should not be included if the labour and material cost is already included.

Table 5.3. Likely health sector and non-health sector food safety costs

Health sector costs	Costs outside health sector
Research	Industry
Setting standards	• Surveillance
Surveillance	• Equipment (e.g. refridgeration)
Legal costs of enforcement	• Legal costs and penalties
Education	Consumers
Poison control centres	• Higher prices
	• Preventive measures

5.2.3 Vector control

Costs of environmental management of vectors not only fall on the health sector, but also other government ministries (environment and agriculture), and farmers. While costs are due mainly to planning and implementation of land use and water management changes, there are also recurrent costs. As those implementing the changes, such as farmers, do not necessarily benefit, it may be necessary to subsidise the changes, and any economic losses that result thereafter (such as change in cropping).

Table 5.4. Likely health sector and non-health sector environmental vector control costs

Health sector costs	Costs outside health sector
Research costs	Other government ministries
Planning and supervision	• Land use changes
Surveillance	• Planning
Education	• Water management authorities
Materials such as polystyrene beans	Farmers
	• Adapt to new practices
	• Change crops
	• Investment in improved irrigation
	• Reduced yields
	Consumers
	• Change water storage practices
	• Make latrines 'safe'

Also, measures to prevent vectors breeding in households may be costly, such as use of yards adjoining houses, and latrines. Project staff may not want to rely on compliance of households in the absence of incentives, and therefore may fund materials as well as education to make the intervention successful. While most households may comply, there are costs associated with non-compliers.

5.2.4 Waste management

Costs of waste management include changes in waste storage, collection and disposal, and costs of avoiding leakage of waste into water courses following disposal. The health sector traditionally has little role in waste management (except medical waste), but its expertise is necessary for planning safe collection and disposal of waste. Most of the costs of waste management are likely to fall on industry (to meet waste management requirements), consumers, as well as the responsible government department (whether local or central). Although local councils may be responsible for collecting and disposing of waste, they are likely to meet their costs from direct charges or increases in local taxes. This responsibility may be handed over to a private company. Also, another cost to society is that people employed informally in collecting and reselling waste and scraps will lose their income.

Table 5.5. Likely health sector and non-health sector waste management costs

Health sector costs	Costs outside health sector
Research costs Planning Surveillance Education	Industry • Changing waste management practices • Equipment requirements • Penalties of not meeting standards Consumers • Increased costs of collection (direct, tax) Local council • Planning • Collection trucks and workers • Purchase of land for landfill or incinerator • Equipment for incinerator • Schemes to employ those living off selling waste • Law enforcement Those living off selling waste • Loss of income

5.2.5 Air pollution

The costs of air pollution control measures fall on those whose activities are directly affected by regulations, including industry, motorists, households (e.g. switch fuel source), as well as the government itself. Some of these costs are one-off, such as the purchase of equipment, while others are recurrent, such as monitoring costs, higher petrol prices and lower engine efficiency. Few, if any, of these costs fall on the health sector. The distribution of costs depends on how behaviour is altered due to regulations or the actions of government to improve the compliance with regulations. For example, motorists may avoid the increased costs of motoring by using public transport. Government may encourage the use of public transport through subsidies. Also, government may provide tax breaks for purchase of ‘green’ equipment or relocation to less densely populated areas.

Few studies have estimated the costs of air pollution control measures. One example was Pearce (1996), who suggests that to reduce SO_2 levels fule gas desulphurization equipment is much more costly than fuel switching and energy conservation, although no data were provided. Also, the Office of Technology Assessment (1989) estimated the cost per tonne of harmful gases using a variety of control measures, including reducing the volatility of gasoline (least costly per tonne reduced) and using methanol to power fleet vehicles (most expensive per tonne reduced). Overall annualised costs of control measures were between US$8.8bn and US$12.8bn. Costs also depend on the level of reduction required. If the least costly cost/tonne methods are used first, then costs will rise per tonne reduced thereafter (see Figure 1 in Krupnick 1993). Johansson (1987) also provides crude estimates of pollution abatement costs.

Table 5.6. Likely health sector and non-health sector air pollution control costs

Health sector costs	Costs outside health sector
Research	**Industry**
Setting standards	Pollution abatement equipment required now
Monitoring	Increased operating costs
	Higher prices of future ‘green’ equipment purchased
	Self-monitoring
	Penalties or fines for not meeting standards
	Government
	Pollution abatement equipment
	Enforcement of regulations
	Monitoring
	Subsidise public transport
	Loss of tax revenue due to tax breaks
	Consumers
	Unleaded petrol
	Vehicle inspection fees and consumer time
	Additional items for car
	Increased road use charges (toll road, car tax)
	Change in transport use
	• Inconvenience and increased travelling time

5.2.6 Climate change and stratospheric ozone depletion

Costs of reversing climate change and stratospheric ozone depletion are essentially the same as those associated with reducing air pollution, although there may be additional longer term costs of reducing the impacts of climate change when they occur (such as moving populations when flooding occurs).

5.2.7 Occupational safety

Occupational safety and health costs result mainly from regulatory measures, such as involving safer handling of toxic materials, new equipment, changes in working procedures, regular maintenance and safety checks, administrative costs, and longer breaks for staff. Therefore the bulk of costs is met by employers, although there are public sector costs associated with monitoring, research and setting regulations. The health sector may be involved in providing more regular medical check-ups at work and in educating management and factory workers about aspects of safety. Those costs met by employers are likely to be passed to consumer in the form of higher prices.

Table 5.8. Likely health sector and non-health sector occupational safety costs

Health sector costs	**Costs outside health sector**
Input to regulations	**Industry**
Continued surveillance and monitoring	Investment in safety and health equipment
Doctor time giving medical examinations	Additional investments in capital goods
Health promotion activities	Additional costs of substitution products
(exercise, food, smoking)	Purchase of personal protective equipment
	Additional costs for changed working procedures
	Additional costs for increased maintenance
	Reduced productivity of workers due to breaks
	In-house preventive services, administration, meetings, training
	Other government
	National level infrastructure, inspection, registers

(adapted from Mossink et al 1998)

5.3 Discussion and conclusion

The examples from the literature make it clear that many or most of the costs of environmental health interventions fall outside the health sector. However, this does not diminish the role of the health sector in defining safety standards, overseeing implementing, and monitoring compliance with the regulations, so that the interventions give maximum benefit. This chapter has addressed whether non-health sector costs should be included in CEA, but the issue remains unresolved. As already discussed, to spend society's resources most efficiently, a societal perspective should be taken when deciding which costs (and benefits) to include. However, this is unlikely to happen in a system with limited links between ministries or government departments (as well as industry and consumers), thus leading to a fragmented decision making process. Inclusion of all costs and all benefits does not provide a cost-effectiveness estimate that is specific to health, and therefore it is little use to decision makers within the health sector. Therefore, not until a 'supra-ministerial' level is implemented (such a role is perhaps already played, albeit imperfectly, by Prime Ministers or Presidents) can the societal perspective be taken. While current CEA guidelines do in fact recommend the inclusion of all health intervention costs, whether inside or outside the health sector, it is usually assumed that those falling outside the health sector contribute an insignificant proportion. However, in the case of environmental health interventions the bulk of costs are likely to fall outside the health sector. Therefore, future CEA guidelines should provide more detailed cost classifications, with advice on which non-health sector costs should and should not be included, as well as means of avoiding double counting and excluding 'transfer' payments (such as taxes, fines, etc).

However, the disadvantage of this argument is that it suggests that environmental health interventions cannot be evaluated within the same framework as other health interventions, whose scope is more narrowly defined due to the narrower range of costs and effects. The problem with evaluating environmental health interventions using a different framework to that of normal health interventions is that decisions cannot be made concerning the optimal allocation of resources between interventions that improve or sustain health. *Thus it appears an unresolvable issue, as the optimum cannot be obtained.* On the one hand, using the same framework would disadvantage environmental health interventions, while on the other hand, using a different framework would lead to sub-optimal allocation of health sector resources. Even if a separate ministry was set up to oversee environmental health interventions, the question of what proportion of government money should be allocated between the health and the environmental health ministry would still remain. A possible solution is to introduce well defined criteria which characterises a given CEA, so that a range of outcomes based on different boundaries can be produced. The choice of what should and should not be included then becomes a political one.

6. VALUATION OF BENEFITS

6.1 Evaluation of alternative methodologies

Valuation means the measurement of a variable in monetary units to represent its economic value. As already discussed in Chapter 4, in evaluating the cost-effectiveness of health interventions, the analyst may wish to include those benefits both related and unrelated to health. The controversy surrounding which benefits to include was outlined. While these debates cannot be cleared up here, this chapter aims to identify the strengths and weaknesses of different methods for valuing different types of benefit.

In recent decades, the issue of *'if'* and *'how'* to value human health has received increasing attention, and there still remains considerable controversy and disagreement, particularly where markets do not exist. While valuation of all benefits is important for making decisions, not all benefits are possible to quantify (Arrow et al 1996) or the methods for valuation are so unreliable that it may be better to leave in 'natural' (non-money) units. This debate is very important for those involved in CEA, in valuing both health and non-health benefits. Therefore, the following discussion encompasses and is relevant to both.

According to Freeman III (1993) two links must be established in estimating the monetary values of changes in human health that are associated with environmental changes:

1. The link between the environmental change and the change in health status.
2. The link between the change in health status and its monetary equivalent, willingness to pay or willingness to accept compensation.

Freeman III argues that there are two strategies for valuing environmental changes that affect human health. The first strategy is to "develop a comprehensive model of human behaviour and choice in which environmental quality is one of the determining variables. Such models can provide a basis for measuring WTP directly as a function of environmental change." The second strategy is to "deal with the two links separately. Economic values of changes in health status or health risk would be derived first. They would then be combined with independently derived predictions of health changes or risk changes as a function of environmental change." (Freeman III, 1993, page 315). This would be relatively easy where environmental and health impact assessment mainly relate to dose-response curves linked to pollution, but the many environmental health risks faced by developing countries are much more difficult to quantify (see Chapter 8).

Similarly to health benefits, valuing non-health benefits requires first establishment of the impact of the environmental change on human activities, and then valuation using the willingness to pay principal. This process can be applied to many of the benefits identified in Chapter 4, including medical and averting expenditure saved, production, and amenity and non-use values.

The first link, that of evaluating dose-response functions or intervention effectiveness, will not be dwelt on in this section, but will be discussed more in Chapter 8 under 'Uncertainty' (as environmental health interventions tend to be more difficult to evaluate in terms of impact or effectiveness). For the second link, there are several methods for valuing health and non-health benefits of health interventions, using the willingness to pay approach. In the context of health care, willingness to pay is a useful tool to measure the value that individuals and sometimes institutions place on reductions in the risk or actual occurrence of morbidity and mortality, as individual judgements should be used in making judgements concerning changes in their economic welfare (Freeman III, 1993). Similarly, willingness to pay can also measure the economic value of other impacts of environmental change, impacts that are not necessarily linked to the health impact, such as amenity or non-use benefits (Hanley 1993). The following four methods for valuing willingness to pay are discussed further below (Hanley 1993, OECD 1995; Postle 1997):

1. *Conventional market price, or effect on production.* Market prices are used to value the costs/benefits associated with changes in environmental quality. For example, the "cost of illness" approach, or the "replacement costs" (such as damage to assets) approach. This assumes that market price represents the 'opportunity cost' (the measure of value used by economists).
2. *Household production function approaches.* Expenditure on activities or goods that substitute for, or complement, an environmental or health-related good is used to value changes in the level of the good. For example, "avertive expenditures" to reduce morbidity or mortality risks, or the "travel cost method" to enjoy an environmental good.
3. *Hedonic pricing methods, or revealed (implicit) preferences.* Prices paid for a complementary good implicitly reflect the buyer's willingness to pay for a particular attribute or to accept an increased risk. For example, "wage risk premiums", or "hedonic property prices".
4. *Experimental markets.* The public is asked to value non-market goods within a hypothetical market. This is called the "contingent valuation method".

Each of these valuation methods is discussed in relation to their validity, applicability, ease of measure, and generalisability. In each of these subsections, a distinction is made between fundamental problems with the methods (that cannot be addressed) and technical or data problems (which can be addressed with more research and/or data collection).

<u>Conventional market price, or effect on production</u>: Market prices can be used for those changes in activity where markets exist. For example, changes in medical costs can be estimated by aggregating the unit costs of those services for the numbers of services saved. This implicitly assumes that the unit costs reflect the economic value. Therefore charge data will need to be adjusted for the profit. Also, in developing countries especially, there is limited direct charge or cost data from health services. A suggested step-by-step method proposed by WASH (1991) for estimation of the direct

costs of medical care saved following a health intervention is outlined in Table 6.1 (first column).

Market prices are also used in the human capital (HC) approach, where human life and time spent ill or recovering are valued using future expected earnings. The calculation uses approximations of the value of the increased productivity of individuals through less work loss days (WLD) or restricted activity days (RAD). For a person who dies prematurely, the lost productivity estimate is often given as the stream of earnings that the person would have earned if he or she had not died. Estimates of WLDs or RADs can be made for specific individuals in a detailed study (such as epidemiological studies), or 'average' individuals (such as dose-response function studies). The latter is used more frequently, due to simplicity and low research costs. A suggested step-by-step method proposed by WASH (1991) for estimation of the indirect benefits of production loss saved following a health intervention is outlined in Table 6.1 (second column).

Table 6.1. Estimation of direct and indirect costs using market values.

Direct cost estimation	Indirect cost estimation
1. Estimate proportion of those affected at each level of severity of the disease who desire treatment. 2. Estimate the proportion of those desiring treatment who have access to treatment. 3. Specify the process of treatment for each level of severity of the disease (resource use, number of inpatient days, outpatient visits). 4. Estimate the unit costs of resources used for treatment and the side effects for each level of severity of the disease, taking into account the fact that many fixed costs are not affected by reductions in the use of health service. 5. Estimate total treatment costs for each level of severity of disease without intervention. 6. Determine the proportion of costs that can be avoided in the short- and long-run 7. Determine the direct costs that would have been avoided as a result of the health intervention	1. Specify the type of economy for the population of interest. 2. Specify the characteristics of the economy for the population of interest. 3. Specify the family and community structure. 4. Specify the unit of analysis. 5. Specify the desired measures for productivity changes. 6. Estimate the maximum gain in productive time as a result of the health intervention. 7. Estimate the maximum value of gain in productive time. This requires assumptions about which groups of patient would have worked, and decisions about value of time of children and retired people. 8. Conduct sensitivity analysis using alternatives to base case.

(taken from WASH 1991)

The human capital approach can be applied to time savings not associated with health, such as reduced water collection time. However, the human capital approach is perhaps the most difficult and controversial aspect of valuing health effects associated with environmental changes (Freeman III 1993). While it should be implicitly recognised that the human capital approach is only a proxy measure for the value of the rest of a person's life, there are several serious short-comings of the approach (Hanley 1993, WASH 1991). The most serious is that HC does not provide information about what the individual (or government department) would be willing to pay to obtain a given reduction in the probability of loss of life, which is what we should be interested in (Fisher 1981). Other fundamental problems with HC include:

- It does not measure net contribution to society. This would equal the difference between a person's production and consumption. However, such valuation assumes: (a) full employment and no substitutability of labour; however, in an

economy with unemployment, a new employee can be hired and the productivity loss minimised; and (b) a dominant cash economy, where there exist market prices, which is a globally unfair assumption given the size of the informal and non cash economies in many developing countries.
- Human capital ignores non-market activities important to individuals, and the loss of leisure time or activities. It undervalues retired people, children and home makers, giving them a negative value if consumption exceeds production.
- The resulting value for life is highly dependent on the discount rate used, and the higher the discount rate, the lower the economic value for children.
- It does not value pain and suffering, the individual's own well-being and preferences, nor does it account for the sentiments of the community.
- The economic measure of value, *revealed preference*, is ignored.

Problems with the human capital approach that can potentially be resolved include:
- There is considerable uncertainty about the number of days or years the individual actually takes off work. For example, there exists uncertainty surrounding the long term effects of an acute illness on production. It also requires an assumption about life expectancy.
- Productivity estimation does not take into account the declining economic value as people get older (although this can be estimated).
- Work loss days and restricted activity days depend on the individual, and how (s)he responds to symptoms and illness.
- There is a lack of labour market data in many developing countries.
- Data are not highly generalisable between populations and countries, as values are highly dependent on local factors.

In conclusion, use of market prices is generally highly valid when prices reflect opportunity cost, and when they are easy to measure. However, this method is only applicable where markets exist.

<u>Production function approaches</u>: The production function method (PFM) may be applied either to firms producing goods or services, or to households producing services that generate positive utility. The latter involves households combining certain commodities to produce other commodities. For example, a household may react to water contamination by either purchasing water treatment equipment or by boiling water, both of which involve changes in expenditure patterns and the use of time. This behaviour is called mitigative behaviour, or avertive expenditure. The value of an improvement in environmental quality can be inferred directly from reductions in avertive expenditure (AE) (Courant and Porter, 1993). Therefore, if avertive expenditure and environmental quality are perfect substitutes (increase in one leads to reduction in the other), an approximation of the exact welfare effects on households of a change in pollution levels is provided by the associated change in AE. The value of statistical life (VOSL) can be estimated by identifying the associated change in risk of an AE, and multiplying it by the inverse of the change in risk [= (1/change in risk) $\times$ AE]. The extent to which AE saved represents true value depends on whether:

1. Investments in defensive equipment can be reversed.

2. The decision concerning the level of AE may be influenced by whether the individual expects government authorities to make expenditures (thus saving the individual the cost).
3. AE are due entirely to the environmental change of interest.
4. AE generates benefits other than pollution avoidance. For example, double glazing not only cuts noise levels, but also cut heat losses.
5. AE causes discomfort which is not accounted for (e.g. a seat belt).
6. AE includes altruistic motives. AE that includes altruistic motives can been shown to increase the VOSL by up to a half (Jones-Lee, 1985).

Therefore, AE may not capture all aspects of a benefit, or alternatively, it overstates the benefit. It is not widely applicable, but is instead specific to occasions when individuals change their activities to prevent an outcome. It requires surveys of individual behaviour, although this can be done using routine information if the data are detailed enough. Results are likely to be highly setting-specific, due to the many contextual factors that affect human behaviour (e.g. social norms, income, risk perception, penalty size, etc.).

The travel cost method (TCM) has also been shown to be a useful method for measuring the value associated with environmental benefits, although it has not been used (not surprisingly) to value health benefits. The TCM method also suffers from weaknesses, such as whether a journey is for the environmental benefit alone, or whether utility (enjoyment) is gained from travelling as well.

Hedonic pricing: The hedonic pricing method (HPM) identifies environmental service flows as elements of a vector of characteristics describing a marketed good, typically housing. HPM seeks to find a relationship between the levels of environmental services (such as noise levels or suspended particulate levels), and the prices of the marketed goods (houses). HPM has been used to value such things as noise around airports, earthquake risks, urban air quality, amenity values of woodland, and disamenity values of living near landfill sites. In most of these cases, there are clearly several types of benefit, consisting mainly of health effects and amenity benefits, although there may also be non-use benefits such as option or existence value. Several problems exist with HPM, associated with the regression model required to identify multiple influences on value, including large sample size requirements, omitted variable bias, multi-collinearity, wrong choice of functional form, not recognising market segmentation or monopoly positions, not accounting for impact of expected environmental goods, and not meeting restrictive assumptions of the regression model (Hanley 1993).

Another application of the HPM is the valuation of incremental morbidity or mortality risks by identifying wage differentials due to risk differences. The theory is that workers have to be paid a premium to undertake jobs that are inherently risky (or disagreeable) and this information can be used to estimate the implicit value individuals place on sickness or premature death. Thus it measures, albeit inaccurately, an implicit willingness to pay for reductions in risk of death, or willingness to accept increases in the risk of death. The value of a statistical life[26] can be calculated in this way using the equation: [(1/difference in risk) $\times$ difference in

[26] This should not be confused with the value of an individual life, which is infinite (Jones-Lee, 1985).

wage rate]. However, this calculation requires that the only difference between the two jobs is level of risk, that attitudes to risk are identical between individuals, and that labour markets are competitive. It also assumes that people do not take risky jobs for any other reason that they pay more, and that they correctly perceive risks. In recognition of the many factors that influence wages, regression analysis is used with personal and job characteristics as explanatory variables. However, with breakdown in these assumptions, it is unclear exactly what this method measures. Also, even if this method is accurate in one setting, the values obtained from this technique are not transferable between countries (or even industries), due to differences in attitudes to risk and incomes (Dixon et al 1986).

The HPM clearly contains a high degree of uncertainty, and therefore it is not a reliable method for valuing benefits of environmental health interventions. It requires considerable data sets for regression analysis, and these must contain data on all relevant (confounding) variables, so that the regression is fully specified.

<u>Contingent valuation method</u>. The contingent valuation method (CVM) enables economic values to be estimated for a wide range of commodities not traded in markets, such as health and public goods (e.g. clean air and scenery). The technique is now widely accepted by resource economists, following a great deal of empirical and theoretical refinements which took place in the 1970s and 1980s (Hanley 1993). CVM works directly by soliciting from a sample of consumers their willingness to pay and/or their willingness to accept for a change in the level of environmental service flows, in a carefully structured hypothetical market. Bids are then obtained from the consumers, bid curves estimated, and the data aggregated to estimate the market demand curve. A detailed explanation of the CVM exercise is provided in most economic valuation textbooks (e.g. Hanley 1993), and will not be described here. One useful aspect of CVM is that questions can be structured so that respondents value only the benefit(s) of interest. For example, health, amenity and non-use benefits can be separated for the same environmental health interventions.

However, several potential problem areas associated with CVM are discussed briefly:

1. Biased estimates of value. This includes strategic bias when respondents think the answers they give will influence their welfare; design bias in the order and nature of information presented; mental account bias when the true value is overstated; instrument bias when the way bids are collected (e.g. tax) changes the willingness to pay; and hypothetical market bias occurs when respondents cannot imagine themselves in the market situation.
2. Protest bids. If the respondent does not agree with the scenario they may bid zero or a very large amount, when in fact their true valuation is different.
3. Survey responses cannot be verified, except through comparison with actual behaviour following survey, which are rarely done. However, most of those that have been done have shown non-significant differences only (Hanemann 1994).
4. Choice of welfare measure. Willingness to accept may not equal willingness to pay, and may be greater when individuals have “risk aversion”, and income constrains the individuals WTP bids compared to the unconstrained WTA. For example, Dubourg et al (1993) show that WTA is significantly greater than WTP for reducing the risk of road traffic accidents.
5. When respondents are given time to think about their replies, they may give different values than if they had to give a immediate answer. For example, in a

study measuring WTP for private water supply in Nigeria, Whittington et al (1992) found that respondents who were given 1-2 days to think about their WTP gave significantly lower bids than those who did not.

6. In the context of administering contingent valuation surveys in developing countries, there are serious constraints that may invalidate research not conducted carefully. For example, questionnaires developed in industrial countries need to be adapted to developing countries. Also, there is a need for trained researchers to administer surveys.

However, there are several advantages of CVM over other valuation techniques:

1. It can take into account non-use values. This includes the utility individuals derive from the existence of environmental goods, even if they do not use it. Non-use is divided into option value (the possibility that the person may want to use it in the future), existence value (the person values the fact that the environmental good exists, irrespective of use), and bequest value (the person wants future generations to enjoy it).
2. It can be designed to include only the variables or characteristics of the market relevant to the objective of the study. For example, it can be designed to include purely WTP for health effects, or it can include productivity effects, expenditure averted, etc.
3. They allow individuals to consider the true cost to themselves of a particular injury or illness.

Also, several studies have shown that CVM results are repeatable, both in terms of similarity in results across different settings, but also using a test-retest methodology. In the context of eliciting WTP for environmental health interventions, CVM has considerable potential, and it has already been used widely, including in developing countries. For example, Whittington et al (1990a) have found CVM to be an appropriate instrument to elicit valuations in a very poor, illiterate population in Haiti, where reasonable, consistent answers were provided. While they found no problem with starting point or hypothetical bias, no conclusion was reached about the extent of strategic bias, as no post-intervention survey was done. CVM is discussed in the context of the different environmental health interventions below.

6.2 Environmental health interventions

6.2.1 Water, hygiene and sanitation

As mentioned in Chapter 3, several valuation techniques have received attention from researchers doing economic studies of WHS interventions, including contingent valuation and revealed preference methods which have been used to value water supply. For example, Whittington et al (1991) compared current vending revenues and costs with operation and maintenance costs of a new piped distribution system. Revealed preference was used to estimate WTP in current market situation (purchasing from vendors), and CVM for valuing WTP of improved (piped) water supplies. These data were used to estimate the monthly prices at which households would be willing to receive piped water estimated.

Using CVM, Whittington et al (1993) found that willingness to pay for sewerage services and water supply depended on four main variables. These were (a) household income, (b) whether the respondent owns the house or is a tenant, (c) how much the household was spending on its existing sanitation system, and (d) how satisfied the respondent was with his household's existing sanitation system. Most respondents bid more for improved sanitation than they presently paid for their existing sanitation service. Whittington et al (1990c) found a relationship between the choice of water service (which had implications for time savings) and a household's implicit value of time, and thus concluded that the choice of technology for a given community will depend on the value of time which households assign to the time savings. Thus these WTP estimates are clearly highly context-specific.

Therefore CVM is clearly useful for eliciting the value of services that are not currently delivered. However, the results from CVM in the water studies did not separate the health from the non-health benefits, and no mention of health impact is made in most surveys. The one exception was Boadu (1992), who found a statistically significant positive relationship between household's history of water-related illness and their WTP for water. Thus households are willing to pay more if they have experience of water-related illness.

Other CVM studies which assessed willingness to pay for quality improvements in water (ground, river, lake, sea) were related more to health effects. For example, bathers at beaches in Portugal and the UK were asked what they value an avoidance of a case of gastroenteritis. A study by Giorgiou et al (1996) asked respondents whose health had been affected to value a gain in health status, and also asked other respondents who had not been affected to value avoiding a case of illness. The payment mechanism they used was annual increase in water rates. Respondents were compared in terms of whether they thought their behaviour changed the probability of them becoming ill from swimming in polluted waters. There were several contextual factors important in determining WTP, including the disease type and severity, the probability of illness, and the personal context (whether they were a holiday maker, day tripper, or resident).

Despite these studies, Machado and Mourato (1999) comment that few economic studies have separated out health benefits from other benefits (amenity) of water quality improvements. In their study, they found that amenity related use values were found to constitute a very high share of the total benefit from water quality improvements. Their WTP survey described gastroenteritis (symptoms, restriction on activities, duration) and respondents were asked to say what they would be WTP to avoid waking up the next morning with it. Also, contingent ranking was used with different values for cost and water quality.

The hedonic price is not as valid as CVM for valuing water supply, due to the many housing characteristics that determine price, although HPM studies have been done. North and Griffin (1993) valued improved water supply using the HPM in the Philippines. WTP for private taps represented about ½ imputed monthly rent, but for deep well connections and public taps WTP close to zero. Also, d'Arge and Shogren (1989) compared property prices between two areas in Iowa with different water quality using the hedonic price method, and found 23% of the difference was due to differences in water quality.

The human capital approach was also used in several studies to value illness avoided due to WHS interventions. For example, the Inter American Development Bank assumed that time savings should be valued at 50% of the market wage rate for unskilled labour in the local economy, although they gave no empirical justification for this assumption. Harrington et al (1989) assumes that the before tax wage is an adequate representation of the social value of lost work. They include both days lost from work as well as lower productivity when a work day is not missed. From a preventive measure, the following costs are saved: direct disutility of illness, lost work productivity due to continuous or intermittent symptoms (before-tax wage), lost work time (before-tax wage), lost leisure (after-tax wage), medical expenses, and defensive expenditures. The latter is change in consumption patterns, such as buying different types of water. This may or may not be successful in avoiding illness. Note however that some averting action costs have a transfer effect, such as purchase of water.

The convenience value of water supply is potentially important. The value of time could be implied from the choice of water source, between vendors (minimum time required), kiosk or collection from open well (maximum time required). However, there are the issues of jointness of activities (whether hauling was combined with other activities), aggravation and inconvenience of water collection, and water quality. The value of incremental activities from time savings has not been measured, and this requires observational methods.

Finally, amenity value of water was suggested by WHO (1997) as deserving more attention. For this the CVM or travel cost method is appropriate.

6.2.2 Food safety

The main benefits of food safety (avoided costs to patient and industry outbreak costs avoided) can be measured using several methods. Market values (actual or implicit) should be the most widely applicable for measuring medical costs, avertive expenditures and production losses, although contingent valuation is the best means of measuring individuals' preferences. However, most benefits are associated with the health effect, and therefore limited consideration is needed of non-health benefits.

6.2.3 Vector control

Market values can be used for most benefits of vector control, including medical costs saved, other vector control activities saved, and increased economic productivity. For example, Audibert (1986) showed that a 10% rise in the prevalence of schistosomiasis resulted in a 4.9% fall in agricultural output (rice) in Cameroon. This occurred through two principal factors: it influenced the area that a family can cultivate, and affects the duration of transplanting of seedlings. Therefore, the economic benefits of the health gain in itself was not valued. However, the economic benefits associated with land use changes have not been included in health studies, and they require microeconomic studies of behaviour and output changes. Also, due to the context specific nature of these benefits, they are unlikely to be widely generalisable. The value of reduced inconvenience or nuisance should be valued using contingent valuation.

6.2.4 Waste management

Market values can be used for some of the benefits of waste management, such as health care costs saved and electricity generation displaced. However, a major benefit of waste management, amenity, should be valued using either HPM or CVM (one study found on each). Existence or bequest values should also be valued using CVM.

6.2.5 Air pollution

Measurement of the economic benefits of reduced morbidity and mortality has been done by a number of studies, using market values for the costs of health care, avertive expenditures, and lost work time (see Table 3.6). For example, Kevin and Weiss (1992) estimate costs of asthma in the United States for 1990 and find direct costs US$3.6bn and indirect costs US$2.6bn.

Measurement of economic value of improvements in air quality is difficult for some benefits not found in markets, such as improved quality of life due to better visibility, less time spent travelling to avoid poor air quality (such as recreational activities outside cities), and non-use values. There are also several benefit associated with reducing traffic (as an air pollution reduction measure), including less noise, less accidents, and land value. For most of these benefits, carefully constructed questionnaires are required to identify the values people place on these benefits, as no separate markets exist for most of these ‘goods’.

Alberini et al (1997) value health effects of air pollution in Taiwan using CVM (avoid recurrence of episode of acute respiratory infection), and several cultural factors are found to determine WTP. However, few studies were found that used CVM for valuing such improvements in air quality. Alternatively, the travel cost method can be used to value amenity, although it is most useful for recreational facilities which have to be ‘travelled to’ for their enjoyment. Hedonic pricing could in theory be used for identifying the value people place on improved air quality, either in terms of their living environment, or their work environment. An example of the former is the study done by Harrison and Rubinfeld (1978) who valued differences in air quality in Boston. They estimated WTP for 1_{pphm} improvements in nitrous oxide levels for households at 3 different levels of income. Their results showed that WTP is correlated with income, and also WTP depends on how bad the present level of pollution is (more means more WTP). No studies were found valuing air quality improvements in the work environment in terms of wage differential, probably due to the mix of factors that determine wages in addition to risk and nuisances, of which air quality is just one.

6.2.6 Climate change and stratospheric ozone depletion

In addition to air pollution, benefits of non-use and future damage costs avoided must be included when valuing climate change and stratospheric ozone depletion. While CVM may elicit some value concerning existence and bequest (which may be considerable), future costs saved may be more difficult to estimate due to uncertainties about the future health impacts and the cost of medical care. Also, with the expected rise in sea level, property prices at sea level are likely to decline gradually over the next few decades.

6.2.7 Occupational safety

Health benefits and associated economic benefits are the greatest bulk of OSH measures, and therefore valuation should focus mainly on those. Mossink et al (1998) find that OHS studies value output loss from the labour market due to sickness absence using a variety of methods. The hourly shadow wage had been valued at the net national product per work hour, average wage rate, marginal cost of labour, and even the net loss to a company due to staff absence (if identifiable, perhaps for small companies). Also, adjustments were made by age and by gender. Some studies valued the loss of household production: one valued it at 100% of the daily wage, while another used the wage of a housekeeper.

Leigh et al (1996) estimated the direct and indirect costs of occupationally-related illness and injury. They categorised medical expenses within injury categories (deaths, permanent total disability, permanent partial disability, temporary disability, one to seven days of disability, non-disabling). They used US$4m for injury deaths and US$40,000 for non-fatal injuries, taken from the low to middle range in the literature. No willingness to pay studies on OSH were accessed in this review, although Leigh et al (1996) review some of the literature.

Other means of identifying economic value of production time lost includes worker compensation claims. However, these may not reflect the cost of illness or individual preferences, as the contents of worker compensation claims varies between case – for example, some include only ‘suffering’, while others include future earnings lost, costs of the victim, or even punishment to the company.

Finally, non-health benefits require less attention than in other environmental health interventions, as the proportion of total benefit is likely to be lower. However, increases in productivity may be measurable according to value added or profits, and job satisfaction can be measured using CVM, although it needs to be separated from health benefits.

6.3 Discussion and conclusion

Table 6.2 presents the recommended methods of valuation for different types of health and non-health benefit (taken from several studies, including Postle 1997; Department of Health 1999). For several benefits, there exists more than one method that can be used, and the table shows the preferred and second best method. While conclusions are provided about which are the best valuation approaches to use for different types of benefit, the reader is referred to the literature for guidance on how to go about valuation itself.

In general, market valuations are the best because they use existing prices and behaviour and are therefore generally valid. However, when markets do not exist, market behaviour must be extracted from surrogate or proxy markets, or from questionnaires. In general, CVM is preferable to HPM as it is more reliable and questionnaires can be adapted to answer primary objectives. HPM requires significant amounts of data to make conclusions about value with any confidence. Household production function is the least applicable, in that it only values averting expenditures or activities in relation to health benefits. However, when avertive expenditure does not have other purposes, the AE method can be used to approximate economic value. Similarly, TCM only measures the enjoyment value, albeit imperfectly, of environmental goods, but may not be very important in valuing benefits of environmental health interventions.

Table 6.2. Recommended methods of valuation for benefits of environmental health interventions

Type of benefit	Market value	Household production fn	Hedonic pricing	Contingent valuation
Health-related benefits				
Improved HRQL				Ö
Improved LE				Ö
Medical costs avoided	Ö			(Ö)
Reduced time spent in care	Ö			(Ö)
Reduced travel expenses to care	Ö			(Ö)
Reduced avertive expenditure		Ö	(Ö)	(Ö)
Increased productivity	Ö			(Ö)
Reduced sick leave	Ö			(Ö)
Benefits not related to health				
Increased competitiveness	Ö			(Ö)
Reduced damage to assets	Ö			(Ö)
Reduced running costs	Ö			(Ö)
Reduced emergency services	Ö			(Ö)
Increased convenience	Ö		(Ö)	Ö
Increased amenity	(Ö)	(Ö)	(Ö)	Ö
Non-use option value			(Ö)	Ö
Non-use existence value			(Ö)	Ö
Non-use bequest value			(Ö)	Ö

Ö **= preferred method; (**Ö**) = second best method**

In addition to the validity and applicability of the instruments for measuring value, environmentalists also have concerns about the use in economics of the WTP approach. While WTP is widely accepted by economists, there still exist several methodological problems associated with conducting WTP studies:

- They assume rational individuals.
- They assume people are well informed about the choices they make.
- Through aggregating values, the preferences of the many are remorselessly outweighing the preferences of the few. This is especially problematic when the majority of people are ill-informed.
- They assume a well functioning market. For example, people are mobile between jobs, and therefore have a choice about trade-off between risk and income.
- Under the cost-benefit analysis system, intrinsic value exists only in humans and not in animals, plants and other natural resources. Therefore, CBA is anthropocentric, and only values non-human entities when humans themselves value them. Put another way, an environmental good that does not enter at least one person's utility function or at least one firms production function will have no economic value (Hanley, 1993). Field (1997) suggests a 'stewardship value', related to the desire to maintain the environment for the continued use of all living organisms.
- Some people disagree with the concept of a VOSL, as lives cannot be treated as 'tradeable' in the same way as goods. Some people may be sceptical of the likelihood that people will be able to express reasoned and consistent choices concerning the value they put on options which increase or reduce the statistical risk of death from various causes (Department of Health 1999).

7. TIME PERIOD AND DISCOUNTING

7.1 Background

Discussion surrounding the discount rate and its role in economic evaluation has been given considerable attention by those working in environmental projects, and is of key interest in projects both related and unrelated to health. Most recent texts on economics and the environment contain a chapter on discounting (Johansson 1987, Hanley 1993, Beckerman 1995, Field 1997). While the rationale for discounting is clear[27], OECD (1995) point out that discounting is potentially damaging to the environment through two routes: first by reducing long term damage to the environment to insignificance, and second by disadvantaging projects that yield environmental benefits in the more distant future compared to those with short term benefits. Therefore the use of high discount rates hastens the rate of exploitation of (non)renewable natural resources (OECD, 1995). In a similar vein, Weitzman (1998) complains that the use of standard discounting methods seems wrong for evaluating projects[28] whose effects will be spread over hundreds of years, such as in the case of disease eradication (e.g. smallpox).

In the environmental health literature, a similar concern is voiced, in that interventions designed to reduce the adverse impact of environmental conditions on health generally involve high start up costs and benefits are not realised for many years, thus making them especially sensitive to discounting. For example, Baldwin (1983) argued in the context of rural water supply projects “the process of discounting removes from consideration a higher and higher proportion of values that fall in the future”. Therefore, the comparison of cost-effectiveness ratios of environmental health interventions with curative interventions leaves the former disadvantaged for two reasons:

- Curative interventions have more immediate effects than environmental health interventions, and therefore a positive discount rate has greater effect on the latter.
- The cost of low technology curative interventions are spread into the distant future, with limited or no start up costs, while the bulk of the costs of environmental health interventions are incurred early in the life of the project. Therefore, a positive discount rate reduces the relative costs of low technology curative interventions.

The general implication is that positive discount rates mean that diseases experienced by future generations have virtually no importance in the present time period (the impact of the discount rate on the net present value of future income streams is shown in Table 7.2). Conversely, a positive discount rate makes preventive interventions worth less, with the outcome that sustainability is a low priority. Discounting benefits tends to give lower priority to projects that have long lasting health effects. For example, the benefit of reducing air pollution in heavily polluted cities is worth very

[27] Individuals discount the future because (a) they may expect to be richer in the future, (b) there is risk attached to investment, (c) people prefer present to future consumption. A full explanation is given elsewhere (Field 1997, Hanley 1993, Beckerman 1995).

[28] For example, global climate change, radioactive waste disposal, loss of biodiversity, thinning of stratospheric ozone, groundwater pollution, mineral depletion, etc.

little if the effects are not felt for twenty or thirty years (the time period it may take regulations to be effective).

7.2 General determination of discount rates

Weinstein et al (1996) state that a convention is required for choosing a discount rate. The discount rate should be set to reflect the current generation's views about the relative weight to be given to benefits and costs occurring in different years (Field 1997). This is represented by the real interest rate (nominal interest rate minus inflation rate), which shows the value gained through time of not spending money now. However, even a brief look will show that there are dozens of different interest rates in use at any one time – rates on normal savings accounts, certificates of deposit, bank loans, government bonds, and so forth. Which rate should be used? There are essentially two schools of thought on this question (Field 1997).

1. The *time preference approach*, which should reflect the way people think about time, such as the average bank savings account rate in a freely operating market (although most private banks are influenced by national banks and politics). Arrow et al (1996), for example, argue that the rate to discount the future should be based on the rate at which individuals trade off present for future consumption. There are several weaknesses of this approach. First: is the notion of impatience, which underlies a positive rate of time preference, ethically defensible? (Field 1997). While such impatience is observed at the individual level, there are questions about whether it should exist at the society level. One reason individuals have positive time preference is that death is inevitable, whereas this is not true for societies. Moral philosophers have questioned whether time should affect the value of a good. If no, then such impartiality implies that the well-being of one generation should not be counted differently from that of any other. The threat of global warming, for example, could present a case for the overruling of personal preferences in favour of future generations. Second, research has shown that short-term, medium-term and long-term savings rates can be significantly different.
2. The *marginal productivity of investment approach*, which is based on the notion that funds used in public sector investments should have the same opportunity cost of capital as those in the private sector. Private sector productivity is reflected in the rates of interest banks charge their business borrowers. This rate is usually higher than the rate of time preference, due to distortions in capital markets. The argument in favour of the use of this rate is that abatement costs tend to crowd out private investment, as most of the costs of abatement fall on the private sector. However, in developing countries where capital resources are particularly at a premium, the opportunity cost of capital may be 15%, which most economists would agree is far too high for use as the discount rate.

The choice of concept on which to base the discount rate has important implications for the cost-effectiveness of health interventions, as the time preference approach can be considerably greater than the marginal productivity of investment approach, and therefore those in favour of a low rate may chose the time preference method and still have a theoretical basis to back up their choice. Often governments set a discount rate to be applied in all public projects, a rate that gets the most support – neither too low nor too high.

However, practices have been shown to differ between agencies in the public sector in the USA, with discount rates between 2% and 10% commonplace, while most agencies support the use of sensitivity analysis. Hanley (1993) resignedly accepts that ultimately the choice of discount rate is a political one, a choice that cannot be resolved in the academic debate alone.

7.3 Discount rates and cost-effectiveness of environmental health interventions

Before solutions are proposed, the potential or actual impact of positive discount rates on the cost-effectiveness or cost-benefit of environmental health interventions is assessed. Table 7.1 shows approximate time periods of the costs and benefits/effects of the different environmental health interventions. In general, most of the costs of these interventions occur in the short term, because they involve compliance with regulations or investment in infrastructure, and the recurrent costs in proceeding years are a small fraction of the initial cost (bold print indicates the bulk of costs or benefits/effects). On the other hand, most of the effects of many of the environmental health interventions occur in the medium to distant future, although this varies between intervention. For example, for WHS interventions, the primary health effect is immediate, and greater health effects are likely over time as societies develop and hygiene and sanitation practices become more ingrained. At the other extreme, it will take many decades or centuries before the full effect of greenhouse gas emission reduction are realised, and even then the actual effect is highly uncertain. Other regulations may take several years or a couple of decades to implement, such as air pollution reduction measures, or occupational or food safety regulations. Some interventions are highly dependent on the infrastructure available, and thus in many poorer countries strict regulations such as those operating in developing countries may not be easily implemented in the short or medium term,

Another aspect is whether the changes are permanent or whether interventions need to be repeated. Also, interventions may have minimal incremental cost. For example, changing emission regulations means companies must invest in new equipment, but further capital costs may not be greater than they would have been in the absence of the intervention. On the other hand, the hardware for WHS interventions may need replacing, which involves a further additional cost to the scenario of no intervention. However, with positive discount rates, further investments (say, in 10 years) will appear small in net present value terms. For example, Varley et al (1998) used a 3% discount rate to annualise WHS investment and operational costs, and assuming 5-10 year infrastructure lifetime. To maintain consistency, they also used a 3% discount rate in DALY calculations.

Table 7.1. Time period of health costs and benefits of environmental health interventions.

Intervention	Present (short-term <5 years?)	Future (long-term >5 years?)
WATER, HYGIENE, SANITATION		
Costs	**Capital outlay**	Recurrent costs (supply, maintenance)
	Set up permanent water supply	**Replacement of hardware**
Health impact	Reduced morbidity and mortality	**Compounded reduction in MB and MT**
Costs saved	Medical care costs saved	**Medical care costs saved**
	Avertive expenditure saved	Avertive expenditure saved
FOOD		
Costs	Regulations and research	Surveillance
	Additional staff for quality control	Staff for quality control
	Capital outlay (equipment)	Update capital equipment
Health impact	Reduced morbidity and mortality	**Compounded reduction in MB and MT**
Costs saved	Medical care costs saved	**Medical care costs saved**
	Avertive expenditure saved	Avertive expenditure saved
VECTOR CONTROL		
Costs	**Cost of land use change**	Maintenance
	Costs of chemical control	
	Cost of home improvements	
Health impact	Reduced morbidity and mortality	**Compounded reduction in MB and MT**
Costs saved	Medical care costs saved	**Medical care costs saved**
	Avertive expenditure saved	Avertive expenditure saved
WASTE		
Costs	Cost of land use change	Running costs
	Capital outlay (trucks, incinerator)	Maintenance
Health impact	Reduced morbidity and mortality	Compounded reduction in MB and MT
Costs saved	Medical care costs saved	Medical care costs saved
	Avertive expenditure saved	Avertive expenditure saved
AIR		
Costs	Regulations and research	Surveillance
	Capital outlay (equipment)	Running costs and maintenance
Health impact	Limited short term impact	**Reduced morbidity and mortality**
Costs saved	Medical care costs saved	**Medical care costs saved**
	Avertive expenditure saved	**Avertive expenditure saved**
CLIMATE		
Costs	Regulations and research	Surveillance
	Capital outlay (equipment)	Running costs and maintenance
Health impact	No impact	**Reduced morbidity and mortality**
Costs saved	Medical care costs saved	**Medical care costs saved**
	Avertive expenditure saved	**Avertive expenditure saved**
OCCUPATIONAL SAFETY		
Costs	Regulations and research	Surveillance
	Capital outlay (equipment)	Running costs and maintenance
Health impact	Reduced morbidity and mortality	**Reduced morbidity and mortality**
Costs saved	**Medical care costs saved**	**Medical care costs saved**
	Avertive expenditure saved	Avertive expenditure saved

(in bold type means that these costs or benefits constitute the bulk of costs and benefits)

Many OSH interventions are disadvantaged by discounting because many of the injuries (such as repetitive strain injury) and illnesses (such as cancer) may take considerable time to develop, and therefore it may take many years before the effects of an intervention become apparent. This is particularly problematic because 10 to 15 years is well beyond the time horizon of most managers. Mossink et al (1998) claim that this has the effect of biasing prevention initiatives towards the more immediately-avoidable events, and disadvantages long-term investments in reducing occupational injury and disease..

Other benefits that are experienced by all interventions, such as increased productivity, will also be relatively smaller for environmental health interventions because future income gains are discounted (due to more distant future health effects). Table 7.2 below shows how the net present value (NPV) of future income streams is drastically reduced with higher discount rates. Also shown is how the NPV of income of children is very small at higher discount rates, as they will not become productive for many years. These data illustrate the potential impact on future events of discount rates, whether they be costs or benefits (costs saved, health gain), and possibly motivates the search for solutions that treat environmental health interventions ‘fairly’.

Table 7.2. Net present value of future income streams for different age groups and discount rates

Age group	2.5%	6%	10%
1-4	405,802	109,368	31,918
20-24	515,741	285,165	170,707
40-44	333,533	242,600	180,352
65-69	25,331	21,807	18,825

(taken from Landefield and Seskin 1982)

7.4 Possible solutions

The problems associated with using positive discount rates for environmental programmes with long-run impacts are difficult to resolve. OECD (1995) and others have argued that those concerned with the effects of a positive discount rate may propose various solutions. The first of these is a *low or zero discount rate*, argued by Cline (1993) for projects with long term effects, as suggested by differences in short and long term interest rates. The Economist (1999) argues that this seems to accord well with the way people think, as many studies by economists and psychologists have found that people do in fact discount the distant future at lower rates than they apply to the future. Also, Drummond and Jefferson (1996) suggest the use of a 0% discount rate in sensitivity analysis, so as not to penalise preventive health programmes. While this solution would indeed increase the relative value of environmental health projects, it has several drawbacks:

- A low or zero discount rate distorts capital markets where governments and private investors are active in the same sector.
- A low or zero discount rate encourages the use of capital intensive schemes, thus discouraging employment, and increasing poverty, which itself puts pressures on the environment.
- A low or zero discount rate would mean unproductive schemes or even highly disruptive public infrastructure projects would be passed, thus increasing the use of natural resources. Norgaard and Howarth (1991) have called this the conservationalist’s dilemma.

Arrow et al (1996) argue that benefit-cost analysis applied to environment, health and safety regulations should take into account equity considerations within and across generations, thus in effect reducing the discount rate closer to zero. However, they also admit that reducing the discount rate purely for this purpose is neither an effective nor efficient tool for achieving redistributional goals.

The second solution involves the *application of lower discount rates to environmental projects or specific environmental effects.* However, the problem with this solution is defining how "environmental" projects would be distinguished from non-environmental projects. Several authors (Pearce and Turner 1990, Markandya and Pearce 1991, Birdsall and Steer 1993) have argued that it is essential for uniform social discount rates to be applied across all investments and that other ways of incorporating or avoiding long term effects and externalities should be used, such as the application of special taxes and subsidies.

The third solution involves *applying distributional weights to costs and benefits accruing to future generations.* However, this involves several philosophical, moral, economic and practical problems, and is a highly arbitrary process. A fourth solution is to discount future health effects at a lower rate than monetary costs and benefits. Parsonage and Neiberger (1992) have argued that the usual rationale for discounting does not have the same meaning when used to justify discounting health effects. Time preference for health has been shown in several studies to be zero or even negative. Also, health has been shown to be more important at different stages of life. The problem with this solution is that not discounting health effects has the result of delaying projects: not discounting benefits at the same rate as costs means that cost-effectiveness increases through time. A second problem is that, as Weinstein et al (1996) point out, health and wealth can be directly traded, which means they should be discounted at the same rate.

A fifth solution involves *leaving health and environmental effects out from development project appraisal,* as they can be classified as 'externalities'. While in the past economists have avoided having to measure health and environmental effects at the macroeconomic level (using partial equilibrium analysis), it is unacceptable to do so these days, and therefore this solution is not recommended.

7.5 Discussion and conclusion

It is not possible in this report to suggest a single discount rate that would not attract severe criticisms, and therefore succeed where many dedicated environmental economists have failed. However, at least some clear suggestions are needed that take into account the many concerns voiced about the negative impact of high discount rates.

First, it is important for analysts and decision makers to be aware of the extent to which discount rates make environmental health interventions with high short term costs and high long term effects less "competitive" compared with other health interventions. Therefore, all options should be discounted at the rate suggested in CEA guidelines for the sake of consistency, but also at zero percent. If possible, the positive discount rate scenario should not be called the base case analysis, so that decision makers do not think this one should automatically be used in choosing health interventions.

Second, as several economists have pointed out, environmental projects should not have the benefit of a low discount rate when other health interventions do not. On the other hand, if there is theoretical and empirical backing that the social rate of time preference is lower for the distant future, then there is little reason why this could not also be presented in a sensitivity analysis, or even as the base case analysis if this proposition can attract sufficient support.

Third, while in the base case analyses both the usual discount rate and 0% should be used, environmental health interventions can indeed be made more cost-effective by being fully comprehensive in its measurement of benefits, many of which are not enjoyed to the same degree for curative health interventions. This approach has been recommended by many environmental economists, and is fully acceptable, although health policy makers may not be interested in many of the costs and benefits. Therefore, health economists working with environmental health interventions would be well employed to work on measuring willingness to pay (when not already available from market values) associated with non-health benefits, when these benefits are to be included in the cost-effectiveness ratio (or will be provided alongside the CER for use in decision making).

Fourth, and the most radical conclusion, suggests that projects with their costs and effects in different time periods should not be compared at all. For example, different ministries should be responsible for the welfare of different generations, or health care interventions should be divided according to whether they are primarily preventive or curative in nature. While these solutions may not be workable in practice, it at least avoids trying to put together a framework that will not gain the support of all players.

8. UNCERTAINTY

8.1 Introduction to sources of uncertainty

The issue of uncertainty and how to deal with it plays an important role in cost-effectiveness analysis, particularly for environmental health interventions. It has received increasing attention since CEA began in the 1970s. Uncertainty stems from a lack of information or a complete or partial lack of knowledge about the consequences of a given action. Postle (1997) lists a number of types of uncertainty:

- Uncertainty over the future relative prices of key cost components.
- Scientific uncertainties (e.g. cause-effect relation).
- Uncertainties regarding the time frame over which benefits will occur.
- Uncertainty as to the most appropriate value for use in benefit transfer.
- Uncertainties about the socio-economic consequences of taking a particular action.
- Uncertainties about policy goals and how to weight one decision factor across others.

There are several reasons why uncertainty may be greater in evaluating environmental health interventions. First, evaluating health effects from a change in the natural or human environment is more difficult to do using controlled experiments such as the randomised controlled trial (RCT). Luken (1985) even argues that cost-benefit analysis cannot be applied in areas where there is limited causal information between pollutants and exposure, and exposure and adverse health effects, such as in hazardous waste. Subsequently larger confidence intervals tend to exist around health effects for preventive compared to curative activities, and thus around economic benefits. Second, costs associated with environmental health interventions have also been found to be highly uncertain, especially when they relate to regulatory measures. These include both costs of the intervention as well as resulting macroeconomic effects, such as competitiveness, prices, and impact on gross domestic product. There is even greater uncertainty surrounding these costs and benefits in environmental health interventions than in other health interventions, and to assess these costs and benefits requires greater cross-disciplinary work. Hahn (1996) suggests that more detailed economic studies should be done before regulations are imposed in the USA, and this may even prevent regulations from being passed. For example, the Environmental Protections Agency's 1990 hazardous waste listing for wood-preserving chemical was estimated to cost US$6.3 trillion per statistical life saved.

A third issue regarding uncertainty concerns the problem of non-transferability of results. This is due to the sensitivity of health effects and economic costs and consequences to many of the variables for the cost-effectiveness calculation (prices, income, exposure, dose-response function, weather patterns). For example, Hanley (1992) argues that the benefit of controlling nitrate pollution depends on percolation rates through groundwater, which are highly locale-specific. Therefore, the issue of whether uncertainty can be quantified using a likely range needs to be explored. For example, Luken (1985) suggests the use of worst-case and best-case scenarios for estimating the costs of compliance with EPA regulations in the USA. However, this approach does not attach probabilities to different outcomes, which may be important for policy makers to weigh up the risks of taking certain actions.

As suggested above by Postle (1997) uncertainty can be found in a number of areas of the analysis, including the data used in the cost-effectiveness calculations, as well as methodological uncertainty about choices that have to be made about the CEA framework. Most important areas of uncertainty have already been partially addressed in previous chapters, but this chapter categorises types of uncertainty, suggests methods for quantifying and representing uncertainty, and discusses the likely levels of uncertainty in cost-effectiveness data for different environmental health interventions. As will be discussed later, sensitivity analysis is proposed as a partial solution, as it allows conclusions to be made about the sensitivity of the results to changes in the values of one or more variables[29].

8.2 Data uncertainty

Under data uncertainty, four areas of particular importance are discussed in this section: cost valuation, effectiveness measurement, benefit valuation, and data transfer.

Cost valuation: Cost valuation is a potential area of considerable uncertainty because data on costs for environmental health interventions are sparse, and methods must be

[29] Sensitivity analysis improves the understanding of an issue by showing more clearly how changes in one variable affect others, it reduces the risks associated with an action by suggesting areas where particular precautions should be taken, and it identifies key variables in the analysis and thus where future research would be best targeted (Postle 1997). If large variations in the assumptions or estimates underlying an analysis do not produce significant alterations in the results then one would tend to have more confidence in the original results (Drummond et al 1997).

identified for getting hold of reasonable data, or making acceptable assumptions. Also, there is often uncertainty related to oversimplification using cost models, and using data that was not collected for the primary purposes of the research study as it is not necessarily known what the data contain. For example, in providing WHS services, the cost per household per year was approximated using average data on cost per capita and multiplying by the average household size (Varley et al 1998). However, the calculations did not take account of the conditions of the setting in question, such as population density, housing type, or prices of hardware and software components. Another example is that of air pollution abatement. Short term costs depend on compliance with the measures, changes in behaviour, while long term costs are even more uncertain, as they depend on the success of the abatement measures, and future technology (Bartonova, 1997). However, costs are easier to collect for some environmental health interventions, when certain data are collected routinely. For example, OSH measures may have data collected on changes in accidents and illnesses, sickness payments to employees, lost production, and changes productivity. In developed countries, where plenty of information exists on the costs and benefits of compliance with occupational safety regulations, this should not be a problem. However, in developing countries such information is likely to be much more limited.

<u>Effectiveness measurement:</u> Effectiveness measurement is one of the areas of greatest uncertainty with respect to environmental health interventions, because of the many assumptions that are usually made in estimating health benefit. Briscoe (1984) argue "an assessment of the likely impact of water supply and sanitation programs on health is far more problematic than the assessment of other components of primary health care which operate more directly on the causes of disease. Thus while it is a relatively straightforward (although not trivial) task to calculate the effects of a tetanus or measles vaccine on death rates, a similar assessment of the effects of a water supply and sanitation program is fraught with methodological problems (Blum and Feachem 1983) and great caution should be exercised in interpreting the results of such studies." Fisher (1981) also argues that it is difficult to estimate the effects of pollution on human health because controlled experiments cannot be carried out on human populations in the same way as they can in plants or mice. Thus the researcher must rely almost exclusively on statistical analyses of public health data, or micro studies of biological function under different environmental conditions (such as the work of Lave and Seskin 1977).

Unfortunately, the problems cited above are not unique to WHS interventions and air pollution control measures, but is generally experienced by all environmental health interventions. Fahs et al (1997) discuss the difficult methodological challenges to estimating the health effects of occupational hazards, and the associated health gains with removal of those hazards. This is partly caused by fragmented information systems, such as the loss of continuous medical histories and follow up data in the USA due to the separation of Medicare from private employer-based insurance information systems. Also, some data are crude or used crudely, as in the case of stage of severity in cancer and approximated transition time, and when this is the case sensitivity analysis is recommended. Reversing global climate change and stratospheric ozone depletion present a particular challenge for cost-effectiveness analysis, due to the uncertainties involved not only in the general impact of efforts to reverse the process, but also the health impact via illness and mortality avoided compared with what would have happened in an alternative scenario.

In order to overcome these difficulties, alternative non-experimental (but weaker) methods have been proposed to measure the likely effectiveness of an environmental health intervention. For example, in their cost of illness study, WASH (1991) propose six stages in estimating the health effects of an WHS intervention programme[30]:

1. Identify the type of disease(s) being targeted.
2. Estimate how the population of interest is currently affected by the disease(s).
3. Determine how other diseases and nutritional status in the population might interact with the disease(s).
4. Specify for the population of interest the expected impact of the health intervention on the disease(s), at different levels of severity.
5. Estimate the number of days/years of healthy life that will be gained due to the intervention taking into account the interactive effects with other diseases and nutritional status.
6. Estimate the long-term effects of the health intervention.

However, the lack of control in this approach severely weakens the robustness of the results, and therefore data on effectiveness generated by this technique should be interpreted with caution. For example, Blum and Feachem (1983) list several methodological problems of measuring the impact of WHS investments on diarrhoeal diseases, including lack of adequate control, sample size of one in cluster randomisation studies, confounding variables, health indicator recall, health indicator definition, failure to analyse by age, failure to record facility usage, and analysis by season. Machado and Mourato (1999) also discuss the problems in identifying a dose-response relationship when considering the health risks of different levels of coliforms and streptococci, due to variability in levels between location, different weather conditions/times of day, and characteristics of person (gender, age, health condition, hygiene), all of which affect vulnerability to polluted water.

Uncertainty may also exist when effectiveness data are taken from reviews of studies. For example, Varley (1998) used a review of 65 studies to generate a plausible range for the minimum effectiveness of WHS interventions. Disability-adjusted years of life saved per case and death averted were calculated using data from Murray et al (1996).

There is also a high degree of uncertainty in estimating the effectiveness of air pollution abatement measures. First, it must be decided which air pollutants to include. For example, Department of Health (1999) identified semi-particulate matter, SO_2, O_3, NO_2 and CO. However, they had to exclude NO_2 and CO due to insufficient evidence of health impact. Second, it must be decided which health impacts to include. For example, Department of Health (1999) included all-cause mortality and increases in respiratory hospital admissions. Third, the relation must be established between the exposure and the health impact. For example, Department of Health (1999) admitted that for deaths it was not known whether they were brought forward

[30] However, despite their elaborate methodology proposed above, Paul and Mauskopf (1993) used a simplified methodology to estimate the number of cholera cases that would have been averted in the cholera epidemic of 1991-2. In this study, they simply used official statistics for the number of reported cases in that time period, and supplemented these data with hospital studies to learn about the age and severity profile of cholera cases. However, official statistics probably underreport the number of cases by a significant amount.

by a few days, months or years, and for hospital admissions it was not known whether people would have been admitted in the absence of the exposure, and if so whether the exposure caused them to be admitted sooner than expected.

Due to these uncertainties, there have been attempts to establish a set of steps in tracing the link between air pollution and health (Dubourg 1995, OECD 1995, Department of Health 1999):

1. *Determine the type and volume of emission*. Which pollutants to include? Pollutants must be easily measurable, as well as have a clear link with illness. Also, if other harmful pollutants are present in a predictable way with the chosen pollutant, it will increase accuracy. When trying to assess the proportion of air pollution by cause, either emission inventories, receptor studies, or dispersion models can be used (Seethaler, 1999).
2. *Estimate pollution concentration at relevant points in the atmosphere* (dispersion model). Air quality varies in time and space, according to emissions, weather patterns, topography and stack (chimney) height. Dispersion models are also an important tool. However, these cannot be predicted beyond the short term. Also, air quality monitoring cannot be done in all locations, so averages must be taken at certain representative locations.
3. *Establish the relationship between specific concentrations of pollutant and human health* (dose-response studies). From either toxicological studies, micro epidemiological studies, or macro epidemiological studies. These studies require a baseline rate of appropriate health outcomes in a population. However, several factors may affect the results, including (a) whether a linear or non-linear function is assumed, (b) whether discouninuities exist, and (c) whether a threshold level exists (OECD 1995). Also, what aspects of health are important must be decided. First, health impacts should only be included that are linked to air pollution. If more than one health outcome is used (e.g. respiratory, cardiovascular, acute bronchitis), they must not be highly correlated, otherwise there will be double counting. Second, how can death caused by air pollution be estimated, and how many days is death brought forward by? The transferability of dose-response functions depends on the starting dose and morbidity levels of population.
4. *Define the population at risk*, and distinguish between 'vulnerable' and not vulnerable (depending on age, location, predisposing factors/illness).

Using this procedure, Pearce (1996) gives the following formula to estimate the economic cost of pollution: [change in ambient concentration × risk factor × stock at risk × unit economic value (cost per death) = total economic cost of production].

<u>Benefit valuation</u>: As already discussed, benefit valuation is an integral part of CBA, and can also be important in CEA. Despite this there remains considerable uncertainty about optimal methods for benefit valuation, especially as the balance must be found between improving data quality and containing research costs. As discussed already in Chapter 6, the data source chosen for benefit valuation depends on the availability and quality of data. Where markets exist, it is usually preferable to use these values. However, for environmental health interventions it was found that many of the benefits are non-market in nature, or require detailed study of behaviour change at the individual level. Because the evidence is mixed for the quality of different research studies, there is uncertainty associated with many benefit valuations. As in the case of costs, uncertainty is increased when models or aggregate data are used, and when

routine reporting sources are used instead of primary research studies. For example, for a national survey of the economic benefits of OSH interventions, the average wage and average medical costs are applied to the average 'case' (severity, age) (Fahs et al 1997).

<u>Data transfer</u>: Data transfer is a relatively new area of research, both for costs and benefits. It involves applying a data set developed from one particular setting to a quite distinct alternative setting. It can be on the epidemiological level (such as transferring a DRF) or the economic level (such as transferring a VOSL). There are several circumstances in which secondary data sources may be preferable or even necessary, including (Hutton 2000):

- Cost data are not always available from routine data systems, due to poor organisation by hospital management, and setting up routine cost data collection systems is costly in itself, and may not have uses beyond the research study.
- Costs are difficult to estimate in the early life of a new health intervention, when data are unavailable or they do not reflect costs that would pertain after wider implementation.
- An immediate decision needs to be made, and collection of primary data would take too long.
- The types of health care for which cost estimates are required may not be provided by the health system at present, and therefore costs cannot be estimated based on current health care processes.
- Cost data collection alongside clinical trials may disrupt health services, and even change provider and patient behaviour.
- There is limited budget for a prospective costing study using primary data. Secondary data therefore allow quick and relatively cheap assessments of health technology before investments are made in evaluating the health technology, and they allow information to be condensed from many sources.

Where it can be shown that values are not likely to differ between setting, data transfer is most valid, and there are many examples where settings are different. For example, the value of environmental vector control is likely to be site-specific, depending on the environment and size and location of breeding sites, and on the degree of malarial endemicity (Mills 1991). Also, transferring non-market values is even more difficult to validate (e.g. Boadu 1992), although it is possible from regression analysis to get a picture of how different characteristics affect WTP, and thus increase the generalisability of results. Alberini et al (1997) compares WTP for air quality improvement in Taiwan to the USA, and found that WTP was lower in a low-income country, but less than proportionally to the income differential.

One type of benefit that is used widely, and thus generalised across settings, is the data on the value of a statistical life. Such data is easier to generate from certain areas, and authors often prefer to use the values of other experts rather than generate their own in their setting. Although it has the disadvantage that peoples preferences are not taken into account *in that setting*, at least the same value can be applied across all public sector interventions, and thus has beneficial equity implications. However, it is possible that WTP varies for different types of risk. For example, because air pollution risks are seen as involuntary and not under the individuals' control, they may value reductions in risk higher than other risks.

8.3 Analytic uncertainty

Analytic uncertainty refers to methodological choices that analysts must make in the course of research (termed the 'reference case' by Gold et al (1996)), and in some cases is linked to data uncertainty (such as choosing the data source). The most important aspects of analytic uncertainty such as variable inclusion and discount rate have already been discussed in detail in previous chapters. However, there must be some agreement on which areas of uncertainty need to be tested further in sensitivity analysis, and how to choose alternative scenarios that have meaning.

The variables tested in sensitivity analysis depend on what are included as the base case analysis. If costs and benefits are fully comprehensive in scope (i.e. societal viewpoint) then one may wish to represent costs and benefits from different perspectives, such as the health sector, patient, public sector, or private enterprise. This provides important further information, which may stimulate interest in cost recovery. If costs and benefits were not fully comprehensive in scope, then the sensitivity analysis should include in the CER data on all relevant (if available). As stated in the previous chapter, alternative discount rates should be used, such as the recommended rate in guidelines, as well as 0%, and perhaps discounting costs and benefits differently, so that all viewpoints are represented.

9. CONCLUSIONS AND RECOMMENDATIONS

It has become clear from the examples and discussion in Chapters 4-8 that several issues arise in the economic evaluation of environmental health interventions that are not adequately dealt with in current CEA guidelines. In general, environmental health interventions have a broader range of costs and benefits, many of these costs and benefits fall outside the perspective of the health sector, a range of valuation techniques may be required to capture the benefits, uncertainty is a potentially significant problem, and some environmental health interventions are disadvantaged when discounting is used. So what conclusions can be drawn from the literature review and discussion of these issues? Conclusions are made for each question posed in Chapter 1.

How can non-health benefits be taken into account in cost-effectiveness analysis to reflect overall benefits of environmental health interventions?
If a societal perspective is taken, all benefits related to the intervention should be included, although these must not be double counted. The main beneficiaries from environmental health interventions include private enterprise, other government departments, consumers, and the health sector. However, for a health sector analysis where decisions are not made jointly with other ministries or private organisations, the cost-effectiveness ratio should only include the health benefits, cost savings and other economic benefits directly associated with the health intervention. The decision maker should also know the relative importance of other costs and benefits of the health interventions. Also, those benefits that are measured should be disaggregated by beneficiary and time period, and whether related to health or not, in order to allow different perspectives to be presented.

How can avoided costs due to prevented illness or avoided disease burden be taken into account in cost-effectiveness analysis?

Avoided costs due to prevented illness are already included in current guidelines, so no specific recommendations are made here. However, due to the time delay of many of the prevented illness, the actual savings may occur in the distant future, thus making the savings highly uncertain, and reducing the value of these at a positive discount rate. Therefore, as recommended below, sensitivity analysis should be done on uncertain variables, and the results under 0% discount rate should be provided as well as at the rates recommended in CEA guidelines. Also, cost savings should be disaggregated by time period, so that policy makers know when the benefits accrue.

How can the willingness to pay of beneficiaries of environmental health interventions be taken into account? Should costs outside the health sector be included in the numerator of the CER?
Willingness to pay for different benefits should be differentiated clearly, as some can be included whereas others should be excluded in CEA. Identification of those willing to pay, and the amount they are WTP, is important from a cost-effectiveness as well as a cost recovery point of view. In economics, individual preferences are those that should count, and therefore WTP of beneficiaries is more important than predicting impact on work days or production. The policy maker will be more interested in those benefits that have immediate economic impact (such as medical costs saved) or those benefits that people are willing to pay for, than those that have non-economic or delayed impact. However, all benefits can potentially be valued in monetary units, and some form of charging scheme set up where appropriate.

Therefore, no final conclusion is drawn concerning which costs and benefits should be included in the cost-effectiveness ratio, as it depends on a number of factors, including (a) the scope and interest of the Ministry of Health, (b) influence of other ministries on the Minister of Health's decision, and (c) whether data are collected on the relevant costs and benefits and valued in monetary units. In conclusion, assuming the analyst has to resources and data to do so, it is recommended that costs and benefits from a range of viewpoints are collected and presented, and the final choice of viewpoint to adopt is left for the policy maker.

How can non-health benefits be valued in monetary units? What are the basic assumptions for their inclusion in cost-effectiveness analysis?
Several methods were identified for valuing both health and non-health benefits in monetary units. Some are widely applicable, such as contingent valuation, whereas others can only be used in specific situations, such as hedonic pricing. While they all have their weaknesses, the most suitable method can be chosen for each type of benefit, and uncertainty in the values expressed and tested in sensitivity analysis. While the literature contains examples of monetary valuation for most types of benefit, the area of benefit that has received least attention is the non-use value. It is recommended that methods are further developed and validated for applying CVM to the valuation of amenity and non-use benefits, as they are particularly relevant for environmental health interventions (e.g. air and water quality, climate change, waste management). Also, the most appropriate methods for measuring the societal gain from indirect benefits of environmental health interventions (such as productivity) need to be decided.

Is discounting appropriate? What rate should be used?

Discounting is appropriate in that it is a clearly rational means of valuing future costs and benefits, given individuals' positive time preference. However, arguments for the adoption of a low or zero discount rate have gained support from among environmentalists and economists alike. Current guidelines recommend discounting at different rates, including a zero rate. However, which rate is used to advise policy makers is clearly critical. The option of using different frameworks for different health interventions may lead to inconsistent choices, but it may be an appropriate compromise solution given the difficulty of applying a single standard framework to the diverse range of interventions that sustain or impact health.

How can uncertainty be dealt with?
Dealing with uncertainty essentially requires information on the extent of uncertainty, ranges on base case values, relationships between variables (correlation), and the conduct of sensitivity analysis (one-way, multi-way, probabilistic). However, the weaknesses of this approach should be recognised, including the fact that a good sensitivity analysis does not make up for poor quality data when better data could have been collected. Therefore, further syntheses of available data and information as well as primary research and field studies are recommended for improving the quality of information for estimating the cost-effectiveness of environmental health interventions.

What lessons can be drawn from the environmental economics field?
The environmental economics field is more advanced than the environmental health economics field in several ways, and has produced acceptable alternative methods or additions to the cost-benefit framework developed during the 1970s. This includes the issues of discounting and inter-generational equity, valuation of non-market goods using the contingent valuation and travel cost methods, measurement of the link between the environment and human activities (including health) including dynamic effects, and the development of practical means of using a comprehensive (societal) perspective. Further work is needed to assess the relevance of these developments for evaluating economic aspects of environmental health interventions.

Therefore, the following recommendations are made, for further discussion:
1. Willingness to pay for non-health benefits should be taken into account in the cost-effectiveness ratio, and subtracted from costs. It should be noted clearly *whom* the actual or potential payment falls on, and *when*.
2. Costs of environmental health interventions not incurred by the health sector should also be included in the cost-effectiveness ratio. Again, it should be noted clearly *whom* the cost falls on, and *when*.
3. Benefits valuation methods need to be reviewed in greater depth in the context of developing countries and health impact if willingness to pay is used to value health and/or non-health benefits in the WHO cost-effectiveness guidelines.
4. Cost-effectiveness estimations should be made with a 0% discount rate, as well as the recommended positive discount rate. This reflects the sustainability objective of the WHO.
5. Several cost-effectiveness ratios should be presented that reflect a range of pre-defined viewpoints (e.g. health sector, industry, consumer, society), and policy makers choose which viewpoint they adopt. In the event that a single viewpoint is adopted, some of the data from other viewpoints can be used, such as informing policy makers of cost recovery options.

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